Ecology, Impacts and Management of Pest Birds

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Abstract

Pests can impact significantly upon the economy, the environment and on human and animal health. However, for pest birds there are surprisingly few studies of these impacts and how to reduce them. The aim of this thesis is to advance our understanding of the ecology, impacts, and management of pest birds using case studies for each of the three main impacts.

In considering *economic impacts* I estimate pest bird abundance, describe a novel method for measuring bird damage, and evaluate the efficacy of lethal and non-lethal methods in vineyards and orchards. Using data across 185 property years, netting was found to be the most effective in reducing bird damage. Shooting was not as effective but was one-third of the cost and had 13% lower damage compared with nil-treatments. Scaring with electronic devices and visual deterrents had no effect on bird damage. Despite their widespread use, lethal methods had limited effectiveness for reducing pest abundance.

In considering *environmental impacts* introduced mallards on Lord Howe Island were used as a case study. Phenotypic characteristics suggest that mallards have supplanted the native Pacific black duck on Lord Howe Island. Management alternatives are evaluated and discussed.

In considering *health-related impacts* wild birds and avian influenza in Australia was used as a case study. Here, the ecology of Australia's Anseriformes, and the epidemiology, modes of transmission, and the factors influencing the prevalence of avian influenza in Australia's wild birds are investigated. Risk profiles to improve the efficiency and relevance of wild-bird surveillance are also provided.

The case studies presented demonstrate that an understanding of a pest's ecology, efficient measures of impacts, and thorough evaluations of surveillance and management strategies are essential for effectively managing their economic, environmental and health-related impacts.

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Author's declaration

Except where clearly acknowledged in quotations, references, acknowledgements and as outlined below, I certify that I am the sole author of this thesis. I further certify that to the best of my knowledge the thesis contains no material previously published or written by another person except where due reference is made below and in the text of the thesis.

Chapter 2 is adapted from the review sections of a published book,
Tracey, J.P., Bomford, M., Hart, Q., Saunders, G. & Sinclair, R. (2007) *Managing Bird Damage to Fruit and Other Horticultural Crops*. Bureau of Rural Sciences, Canberra. Sections on bird scaring, exclusion, chemical repellents, and chemical fertility control were co-written with Mary Bomford. Mary Bomford, Quentin Hart, Glen Saunders, Ron Sinclair, Richard Porter, Peter Fleming, Jim Coleman, Sharelle Hart, Ian Temby, Stephen Eldridge, Leanne Brown, Andrew Woolnough, Laurie Twigg, Gary Martin, Peter Thomson, Marion Massam, Tamra Chapman, Tina Bentz, Brian Lukins, Peter Mawson, Sandy Hathaway, Roger Goebel and Steven McLeod all provided constructive comments

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INTRODUCTION

1. General introduction

‘Pests can be defined as organisms that cause harm: economic, environmental or epidemiological’ (Hone, 1994).

For vertebrates, economic impacts are usually associated with damage to agricultural production, including predation of livestock animals (foxes and lambs: Lugton, 1993; feral pigs and lambs: Choquenot et al., 1997; Europe: Cowan and Feare, 1999); and damage to crops (mice and sunflower: Saunders and Robards, 1983; quelea and grain: Bruggers and Elliot, 1989).

Environmental impacts include competition with other species (invasive, Hemidactylus frenatus, vs native, *Lepidodactylus lugubris*, geckos in Hawaii: Petren and Case, 1996), interbreeding with native species (domestic dogs and wolves: Blanco et al., 1992; Boitani, 1992), or disturbance of the environment (feral pigs in pasture and forests: Hone, 1980; Singer et al., 1984; Hone, 1988). Invasive species can also lead to extinctions (Clavero and García-Berthou, 2005), and can affect ecosystem function (crazy ants on oceanic islands: O'Dowd et al., 2003; Argentine ants in northern California: Sanders et al., 2003). Biodiversity impacts of pests on the environment are difficult to separate from other causes of habitat loss or disturbance (MacDougall and Turkington, 2005). Although various methods can be used to estimate costs of environmental impacts (Sinden, 1994), the true value of the extinction of a species is impossible to determine (Pimentel, 2002).

Epidemiological or health-related impacts include the transmission of infectious diseases involving humans, livestock or wildlife. There are many infectious diseases that originate in wildlife that have substantial impacts on human health, agricultural production, and the environment (Bengis et al., 2004; King, 2004). The increasing emergence of these diseases is usually attributed to international increases in human population, movements and trade (Brown, 2004; MacDiarmid, 2011). However, increased interaction between wildlife, humans and livestock as a result of changes in distribution of wildlife hosts (Bengis et al.,

2004) and changes in agricultural practices (Slingenbergh et al., 2004) may be more important.

Management of wildlife as pests needs to be underpinned by good scientific understanding of the problems. This includes the underlying ecology, but also requires robust methods of monitoring and evaluation, both in terms of quantifying the problems and evaluating success of management. Decisions for managing pests should be made on the basis of the level of impact (Parker et al., 1999) and evaluations of management alternatives (Mumford and Norton, 1984) incorporating uncertainty or risk (Lane and Stephenson, 1998). Adaptive management or 'learning by doing' (Walters and Holling, 1990) allows improved decisions where there is uncertainty (Shea et al., 2002). For pests, adaptive management requires reliable and on-going measures of pest abundance or impact and management effectiveness. Inaccurate measurements of impact can result in poor management decisions. For example, the cost-effectiveness of many rodent control programs in Hawaii are questionable where the 10% damage recorded annually (Tobin et al., 1993) has no measurable impact on the yields of mature nuts (Tobin et al., 1997). In practice measurements of pest abundance or impacts are rarely conducted, or are complex, qualitative, correlative, or anecdotal (Bomford and O'Brien, 1995; Reddiex and Forsyth, 2006; Tracey et al., 2007). For example 67.5% of pest control conducted in Australia does not involve monitoring of either the pest or biodiversity (Reddiex and Forsyth, 2006).

Most valuable information about a species relies on some measure of population size. However, the reliability and cost of many techniques to measure pest abundance and impact are major limitations to adaptive management of pest populations. Pest abundance can be measured as an absolute number or density, or as an index– one population relative to another (Caughley, 1980). Indices or relative measures are often more useful in practice than absolute measures (Ruscoe et al., 2001; Caley and Morley, 2002; Tracey et al., 2005b). There are many techniques used to measure pest abundance including capture-recapture (Pollock et al., 1990), removal methods (Eberhardt, 1969), distance sampling (Barry and Welsh, 2001; Buckland et al., 2004) and aerial surveys (Caughley, 1974; Tracey et al., 2008a). Measures of pest abundance can be used to evaluate the effects of management (e.g. shooting, Saunders and Bryant, 1988; and baiting, Fleming et al., 2000 for feral pigs), or to predict damage (e.g. impact of rabbits on pasture and wool production: Croft et al., 2002; Fleming et al., 2002a).

While for some pests, measuring damage directly can be simpler and more accurate than estimating abundance and inferring impacts (e.g. pest birds in horticulture: Tracey et al., 2007), measuring impacts is difficult and time-consuming. Pest impact can be measured in a range of ways including: monitoring threatened species, measuring predation to livestock, counting, weighing or visual assessment of damage to crops, estimating losses to pasture or crops or predicting risks of disease.

Vertebrate pests cause significant impacts in Australia and New Zealand (McLeod, 2004; Gong et al., 2009). In Australia 33 mammals are regarded pests, 16 of these are introduced; and New Zealand has 25 introduced pest mammals (Cowan and Tyndale-Biscoe, 1997). Costs of managing these species are significant. For example Government conservation agencies in New Zealand spend 20% of their annual budgets on introduced mammals (Parkes et al., 2006). In addition, over 100 bird species are regarded as pests in Australia and New Zealand (Tracey et al., 2005; Tracey et al., 2007), which cause significant costs to agriculture (Gong et al., 2009) and pose unmeasured risks to the environment, and human and animal health.

Most studies on vertebrate pests have focused on mammals (Europe: Cowan and Feare, 1999; North America: Timm and Fagerstone, 2010; Australasia: Saunders and Lane, 2011). While considerable ecological and behavioural research has been performed on birds (Handbook of Australian New Zealand and Antarctic Birds Volume 1-7; Handbook of the Birds of the World, Volume 1-14), relatively little has been done on birds as pests. The aim of this thesis is therefore to advance our understanding of the ecology and management of pest birds, using case studies for each of the three main pest impacts – economic, environmental and health-related. This thesis provides improved, efficient methods to estimate pest abundance and impact, and empirical evaluations of strategies to manage these impacts, in agricultural crops, on native fauna and in evaluating risk of disease. It also demonstrates the importance of incorporating pest ecology in estimating and managing the impacts of pests.

After a review of methods to assess the impacts and control of bird pests (Chapter 2), the thesis is divided into three parts to reflect economic, environmental and health-related impacts. Part A addresses the economic impacts of bird pests and the effectiveness of their

control in reducing damage. Firstly, a technique to measure the damage caused to wine grapes by native and introduced birds is developed (Chapter 3), and then the effectiveness of lethal and non-lethal methods in reducing damage to fruit is measured and contrasted (Chapter 4). The hybridisation of a native species with an introduced Northern Hemisphere anatid (Chapter 5) is used as an example of the environmental impacts of pest birds (Part B). To demonstrate the health impacts of birds (Part C), an ecological framework is used to investigate the potential role of wild birds in the introduction of a zoonotic disease, highly pathogenic avian influenza, which is currently exotic to Australasia (Chapter 6). Chapter 7 is a risk assessment of the potential avian vectors undertaken as an aid to contingency planning and management of avian influenza. The final chapter uses the case studies to provide a synthesis of the three impacts of bird pests in Australasia and makes suggestions for future research.

2. A review of methods to measure and manage pest bird impacts

2.1 Publication bias

When reviewing literature authors need to consider the possibility of publication bias, which is caused by the selective reporting of scientific results (Dickersin 1990; Moller and Jennions 2001). There are various ways to test and correct for this bias (Rosenthal 1979; Moller & Jennions 2001). While the influence of this bias for pest birds is currently likely to be minimal with few rigorous field evaluations available (This chapter; Tracey et al., 2001), future consideration of this bias may become increasing important. For example selective reporting of positive management action may lead to an over-emphasis on active control of bird populations where it is not warranted.

2.2 Methods to measure impact

Appropriate impact assessment is a critical step in the effective management of pests (Hone, 2007). Assessment allows for comparison of control effectiveness and for improved planning and evaluation. The methods used for measuring pest bird impacts include: questionnaires: face-to-face interviews, phone interviews and mail surveys; direct measures to estimate damage to crops; and environmental and health-related impacts; and indirect measures: monitoring bird numbers and energy demands. These are reviewed below. The benefits of birds should also be considered in assessing their overall impact (Section 2.4).

2.2.1 Questionnaires

Questionnaires about damage and control methods used are useful in defining the problem and for setting research and management priorities over large areas. Face-to-face interviews (Bennett, 1984), phone interviews (O'Donnell and Vandruff, 1983) and mail surveys (Atwood, 1956; Dawson, 1970; Crase and De Haven, 1973; Stickley et al., 1979; Wakeley and Mitchell, 1981; Bomford, 1992; Johnston and Marks, 1997; Graham et al., 1999) can all be used to gather damage information (Bryman, 2012; Fleming et al., In Press). There is a trade-off between obtaining specific information and the time and cost involved (Miller, 1983; Crabb et al., 1988). Face-to-face interviews are more useful when complex information from specific groups is required (Orlich, 1979), but they are more

time-consuming and costly than mail or phone surveys. Mail surveys can be used over larger areas and have the lowest cost per response.

All questionnaires have potential biases. For example, biases can occur when a proportion of the targeted sample does not respond (Dawson and Bull, 1970), when the survey is conducted too long after the impacts have occurred (Sen, 1972), or when respondents overestimate or underestimate damage (MacDonald and Dillman, 1968). Other errors can be reduced by carefully wording questions to avoid leading particular responses. Correct and objective phrasing and ordering of questions has been reviewed by a number of authors (Kahn and Cannell, 1967; Orlich, 1979; Filion, 1981; Chadwick et al., 1984; Crabb et al., 1988).

In some cases, biases associated with questionnaires can be corrected (MacDonald and Dillman, 1968; Sen, 1972). For example, fruit growers with significant bird damage may be more likely to respond to a questionnaire about birds (Dawson and Bull, 1970). This 'non-response' bias can be estimated by re-sampling a proportion of the candidates that did not reply. Rankings of damage can be correlated with actual damage determined with direct measures (Martin and Crabb, 1979; Somers and Morris, 2002).

2.2.2 Estimating damage to crops

Without counting and evaluating all plants within a crop, estimation of bird damage requires the taking of a representative sample from which total damage is predicted. Standard random and systematic sampling procedures (Granett et al., 1974; Caughley and Sinclair, 1994) are used to achieve accurate and precise measures. The desired degree of accuracy or precision will determine time and cost required. Direct measures of damage include weighing, counting and visual indices. Counting and weighing are time consuming but can be used to calibrate standardised visual approximations. These techniques have been used for cereal crops (Dawson, 1970; Khan and Ahmad, 1990) and for apples, pears and stone fruits in orchards (Long, 1985). Weighing and counting often fail to account for losses due to secondary spoilage.

The decision to use weighing, counting or visual estimates will depend on the type of crop as well as the available resources. For example, when measuring damage to grapes it is often not practical to count all the individual berries on each bunch, so a visual estimate

may be preferred. However, for larger horticultural crops such as vegetables and stone (such as peaches and cherries) and pome fruits (such as apples and pears), counting may be just as efficient — and more accurate. Where damage is patchy within an orchard block, stratification will increase precision and decrease sampling effort. If sampling is being conducted over larger areas, stratification according to the age of the crop, geographic area, variety, and early or late maturing date can also increase sampling efficiency and accuracy (DeHaven, 1974b).

Weighing

Calculating bird damage by weighing involves cutting off and weighing a representative sample (plot) of individual fruits. This method has been used for measuring damage to grain crops (Khan and Ahmad, 1990). The undamaged weight of a fruit or bunch is calculated from the mean weight of the undamaged samples in the plot. An estimate of the damage in each plot is then calculated from the difference between this weight and the actual weight of the whole sample from the plot. However, in most horticultural situations weighing is impractical because of the variable weights of fruits and failure to take into account pecked and partly damaged fruits or plants.

An alternative weighing method can be used when distinct areas of the crop have been damaged exclusively and are therefore unharvestable. For example, consider several rows of wine grapes that are severely damaged by starlings[+] (*Sturnus vulgaris*) to the extent that they have become uneconomic to pick. The weight of fruit or nuts lost from rows not harvested could be estimated from the average weight of harvested fruit or nuts from undamaged rows of an equivalent variety and age. Although this provides estimates quickly, it also assumes negligible damage has occurred in other areas.

Counting

Estimates can also be calculated by counting the number of damaged and undamaged samples within a crop. Although counting has been used to estimate total damage (Burton, 1990), a common use of this method is to calibrate visual estimation methods (Stevenson and Virgo, 1971; DeHaven and Hothem, 1979; Somers and Morris, 2002).

[+]'starling' in this thesis refers to the European starling, *Sturnus vulgaris*.

Visual assessment

Visual estimation is rapid and a widely used method for measuring damage to agricultural crops by vertebrates (Stevenson and Virgo, 1971; DeHaven, 1974a; Dolbeer, 1975; DeHaven and Hothem, 1979; Somers and Morris, 2002). This is achieved by using experienced observers to estimate percentage loss, or by assigning a damage ranking to individual fruits or plants. To improve accuracy, estimates may be calibrated by counting or weighing samples that have been visually assessed. Sample cards or templates containing examples of damage levels can be useful guides for measuring losses visually and standardising between observers (Fleming et al., 2002b).

Measuring secondary damage and compensation

In addition to the direct loss caused by birds consuming fruit, crops can suffer secondary losses through spoilage to previously undamaged fruit from moulds, yeasts, bacteria and insects attracted to damaged fruit. This secondary damage is not easily measured in terms of cost, as it is associated with down-grading of fruit by purchasers, extra staff costs to remove bird-damaged fruit and increased costs for fungicide application. Timing and type of bird damage may also be a factor. For example, when wine grapes are damaged immediately before harvest by birds that peck grapes (such as silvereyes and honeyeaters, Meliphagidae), rather than remove them, disease is unlikely to establish and wine quality is uncompromised. There is a need to record the timing and type of secondary damage, as well as the costs incurred.

In some crops, a certain level of bird activity can be tolerated without any significant impact on final yield. This is because plants compensate for fruit or bud loss by increasing the size of remaining fruit (Stephenson, 1981b, a). Compensation for loss to bird pests is measured by comparing yields of damaged and undamaged plants rather than by calculating the percentage of damaged fruit. In many horticultural crops, remaining buds can compensate for damaged buds. For example, in South Australian cherry orchards, removal of some buds can result in larger fruit, which attract a premium price for quality, and lower production costs by reducing the labour required for picking. That is, fewer large fruits are worth a lot more than an equivalent weight of smaller fruits. Therefore some bud damage may, in effect, be similar to the normal horticultural practice of thinning and may result in economic benefits (Sinclair and Bird, 1987).

Conversely, when birds damage the growing shoots of production plants, the secondary shoots are often less productive and are likely to yield more numerous, but smaller, fruit. Damage to growing shoots can also cause reduced productivity from the tree or vine in subsequent seasons (Rawnsley and Collins, 2003). When peas have their emerging shoots nipped off, mainly by sparrows (*Passer domesticus*), the missing shoot is often replaced by two new ones from the seed, but this causes the crop to ripen unevenly and be downgraded by the processing factory (Porter et al., 1994).

An isolated assessment of bird numbers or damage needs to be considered in the context of the effects of damage on the critical stages of crop development and on final production. For example, Tobin et al., (1993) found that macadamia nut trees compensated for rat damage by producing more nuts, and overall yields were unaffected. In this example there are no economic benefits of pest control.

Woronecki et al. (1979, 1980) found that estimates of primary bird damage to corn were affected by: the state of development of the kernels at the time of damage, the amount of compensatory growth, and the environmental factors that influenced secondary loss. For cherries, a reasonably accurate estimate of bud damage could be achieved by a single estimate just before flowering, as new buds are not initiated after flowering.

In many cases, estimates of direct percentage loss will be sufficient as a basis for management decisions. However, these estimates are likely to be conservative when there is a high percentage of pecked or partly damaged fruit; or overestimated where damage takes place early in the season and compensation is likely to occur.

The most appropriate time to measure damage by birds will vary between crops and situations. For example, damage should be measured as close as practicable before harvest when the majority of damage usually occurs late in the season and all damage is easily identified at this time. The situation is more complex when damage is occurring at different stages of growth before ripening, and when damage early in the season is no longer detectable before harvest. In these circumstances, damage should be measured in separate stages and collate the results to obtain overall damage estimates.

[+]'sparrow' in this thesis refers to the House sparrow, *Passer domesticus*, except when preceded by another word to describe the species.

2.2.3 Estimating environmental impacts

Environmental impacts can be measured by estimating change in abundance or distribution of native animal and plant species at risk, competition (e.g. mynas, *Acridotheres tristis*, with native species in Australia: Grarock et al., 2012) or interbreeding with native species (e.g. anatids in New Zealand: Williams and Basse, 2006) or disturbance of the environment (e.g. little corella, *Cacatua sanguinea*, damage to eucalypts: St John 1991)

Biodiversity impacts of pest animals are mainly attributed to mammalian predators or insects, rather than birds. For example, in New South Wales, Coutts–Smith et al., (2007) identified only three pest bird species as a threat to biodiversity, and none of these were on the mainland. However, pest birds are implicated in causing the decline of native fauna and flora. For example bell miners (*Manorina melanophrys*) are implicated in canopy dieback of eucalypts, as a result of excluding other insectivorous birds and the invertebrate predators and parasitoids (Stone, 1996), and the establishment of mynas in Canberra has been linked to the decline in native species (Grarock et al., 2012).

Impacts of pests on the environment are difficult to separate from other causes of habitat loss or disturbance (MacDougall and Turkington, 2005). These impacts are often not obvious and may occur over considerable time frames (Davis, 2003). Pest species themselves evoluting with their interaction with native species (Mooney and Cleland, 2001). Hybridisation (Huxel, 1999) and introgression between invasive and native species can lead to extinction (Rhymer and Simberloff, 1996). However, establishing evidence of environmental impacts based on cause and effect (e.g. Conroy et al., 1989) (rather than correlations in species abundance) is difficult, and rarely conducted for pest birds.

Birds can also damage native vegetation directly. For example, large roosting colonies of little corellas (*Cacatua sanguinea*) (often exceeding 10,000 individuals) along watercourses of the Flinders Ranges are known to cause significant damage to many mature Eucalyptus spp., particularly river red gum (*Eucalyptus camaldulensis*), but also native pine (*Callitris columellaris*), peppermint box (*E. odorata*) and long-leaved box (*E. goniocalyx*) (St John, 1991).

[+] 'myna' in this thesis refers to the common myna, *Acridotheres tristis*.

Hone (2007) outlines 6 principles for measuring and managing pest impacts for biodiversity. This includes 3 principles for the conservation of a species; population limitation, reintroductions and threshold population, and 3 principles for the conservation of communities; threshold habitats, community effects, multiple pests. These principles rely on estimates abundance or occurrence (e.g. proportion of suitable habitats occupied by a species) (Section 2.1.5).

These principles are useful as they allow us to consider the impacts of pests (as populations and to communities) on biodiversity in different ways, generalising across species, location and time. For pest birds, population limitation and threshold population are relevant when considering the direct impacts of pests on a wildlife species. The impacts of mynas, *Acridotheres tristis*, on native species in Australia (Grarock et al., 2012) or the extent of hybridisation between mallards and native anatids in New Zealand (Williams and Basse, 2006) are some examples. However, hybridisation between native and introduced pests warrants further consideration. Will a hybrid fulfil an ecological function equivalent to native precedcesor? It is important to consider the ecology and population level effects of pests, as well as their individual impacts on wildlife. For example, consideration of threshold habitats and the conservation of eucalypt communities is useful when managing little corella, *Cacatua sanguinea*, damage to red gum (*Eucalyptus camaldulensis*) (St John 1991). While damage to these plant communities are a concern to the community, a causal link to tree dieback has not been established (St John 1994; Voller and Eddie 1995) and tree canopy restoration is possible in the short term in favourable conditions (St John 1994). While further exploration of these principles for pest birds are beyond the scope of this thesis, they provide important context to Chapter 5 which is an investigation of the impacts of Mallard on native fauna.

2.2.4 Estimating health-related impacts

Diseases can significantly impact upon economies (Bennett et al., 2009), the environment (Daszak et al., 2000) and human (Binder et al., 1999) and animal (Gortázar et al., 2007) health. Disease events are increasing in frequency and most originate in wildlife (Jones et al., 2008). These are linked to a range of socio-economic, environmental and ecological factors (Jones et al., 2008).

Estimating the impact of diseases involves an understanding of transmission and the effect of the pathogen on the host (McCallum, 2000). The most accurate way to estimate these impacts is through manipulative experiments where pathogens are introduced (McCallum and Dobson, 1995). However, these are rarely practical. Instead surveillance provides information on prevalence and clinical signs provide information on likely affect on mortality. Alternatively diseases in wildlife are modelled with transmission coefficients (spread from infectious to susceptible hosts), which largely depend on contact rates between individuals (Grenfell and Dobson, 1995) and the infectivity of pathogen (Thrusfield, 1995). Infectivity is usually measured in the laboratory (Thrusfield, 1995), however, field contact rates are often difficult to measure (Caley and Ramsey, 2001) particularly in birds that form large and mobile flocks.

Techniques to estimate impact so far have focussed on estimating impact after it has occurred. However, this often prevents adequate management preparation. Although impacts of diseases can be highly variable an understanding of virus epidemiology and ecology of hosts can be used to assess the likelihood of impacts occurring. This is important in assessing likely threats, and provides a basis for improved surveillance and management of wildlife diseases.

2.2.5 Indirect measures

Bird abundance

Many reviews are available of methods to estimate wildlife abundance (e.g. Lancia et al., 1996; Schwarz and Seber, 1999), and occupancy modelling (McKenzie et al., 2006) is becoming popular for estimating species occurrence (e.g. marsh birds, Rush et al., 2009). A variety of techniques can be used to estimate the number of birds or bird species within a given area i.e density (Bibby et al., 2000). A commonly used method that simply identifies the species present is the 20-minute, two-hectare search commonly used in citizen science (Barrett et al., 2003). An estimate population density or an index of abundance is much more useful than the number of species recorded.

Point counts (Lack, 1954), where the numbers of birds of each species are recorded for five- or ten-minute intervals, is one method used to estimate relative or absolute abundance (e.g. Hutto et al., 1986). These counts are usually recorded after first light, when birds are

most active. Caution must be taken to count birds that are more active in orchards at different times of the day, and to take into account differences in detectability between species. There is also a variety of ways to correct for bias associated with detection, such as using sighting distance to estimate the probability of detecting a bird by an observer (Buckland et al., 2001). This method assumes the probability of detecting a bird declines with distance from the observer and that all birds at the observation point are observed with certainty.

The success of a management campaign can be measured in terms of reduced numbers of pest birds (e.g. Conover and Dolbeer, 2007). This type of information can be used in a cost-effectiveness analysis (Hone, 1994). If changes in bird numbers are being used to evaluate management or control effectiveness, the same measurement methods must be used before and after implementation to enable an accurate comparison (Hone 1994).

Relationship between bird population density and damage

Bird population density can be used to predict bird damage without directly measuring the damage. This can be achieved by using the relationship between density and damage (Figure 2.1). Unfortunately this relationship is rarely known and is often difficult to obtain. Any prediction of damage from the number of birds relies on assumptions about density-damage relationships. There is little published information about these relationships for pest birds. Pest density-damage relationships are rarely simple proportional equations whereby halving the pest density halves damage (Figure 2.1). Measurements of density and damage taken over time need to be assessed to determine this relationship.

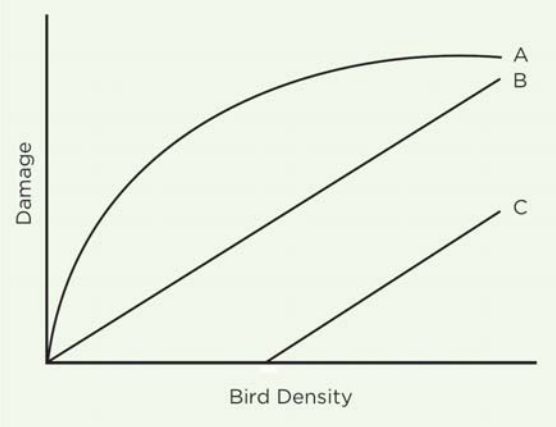

Figure 2.1. Some possible relationships between bird density and damage (After Choquenot et al., 1996).

In Figure 2.1:

- **A** represents a situation where low numbers of birds still cause high levels of damage. Blackbirds[+] (*Turdus merula*) may damage fruit in this way, where a few resident birds can inflict continuous levels of damage over the season.
- **B** represents a situation where damage is proportionally higher when there are higher numbers of birds. This has been shown to occur with bird damage to pistachios, where damage increases directly with increasing numbers of crows per unit area (Crabb et al., 1986).
- **C** represents a situation when damage does not occur until birds reach a certain threshold density. This could occur, for example, when native honeyeaters exhaust a preferred native food source before damaging fruit. This relationship could also occur if cherry trees compensate for a certain level of bud damage by rosellas before production yields are reduced (Sinclair and Bird, 1987).

Even if these relationships are determined, they may be applicable only to a specific situation and often cannot be generalised. Unlike urban and environmental impacts of birds, direct measures of bird damage in agriculture can be simpler, less time-consuming and more accurate than estimating bird density and inferring the impact.

[+]'blackbird' in this thesis refers to the European blackbird, *Turdus merula*, unless preceded by 'American' or another word describing the species.

Estimating bird damage from energy requirements

Information on feeding and energy requirements of target species can also be used to estimate potential impacts. These methods predict damage by translating bird abundance and daily energy requirements of individual birds (Kendeigh, 1970) into the amount of the resource removed.

Bird damage can be estimated as:

> Daily amount of crop consumed = [number of birds] x [daily energy requirements of individual birds] x [proportion of energy obtained from the crop relative to all items consumed] x [energy available per weight of crop].

For example if we have 10 birds, each requiring 70 kilojoules of energy per day, and half the energy comes from grapes, which have 2.15 kilojoules of energy per gram, then the total weight of grape consumed by the ten birds is:

> 10 birds x [70 kilojoules/(day bird)] x [1/2] x [2.15 kilojoules/gram]
> = 163 grams/day

If these 10 birds were of species that remove whole grapes, then the 163 grams/day is approximately equivalent to 163 grapes (wine grapes average approximately 1 gram each) and this is an estimate of the loss. However, if the 10 birds were of species that only peck grapes, the 163 grams may come from many more than 163 grapes and thus under-estimate damage.

More complex approaches using other determinants of energy (such as age class, annual and daily change in abundance and behaviour, temperature and body weight) have been used to predict damage to corn and grain crops by starlings and American blackbirds (Icteridae) (Wiens and Innis, 1974; Wiens and Dyer, 1975; Weatherhead et al., 1982; White et al., 1985).

Considerable ecological information is required for energy and density measurement methods. This requires long-term research, which in most cases is not available (Otis, 1989). Methods using energy requirements also do not take into account the natural variation in damage (Otis, 1989; Hone, 1994); nor do they take into account losses due to secondary spoilage. Despite these difficulties, an enclosure study of American blackbirds

and grain found that estimates of damage using energy requirements and bird density were equivalent to direct measures (Weatherhead et al., 1982).

Estimates of damage using energy are more useful when estimating damage over broad agricultural areas: for example, when density and feeding habits are already known, easily obtained, or being determined for other reasons. When applying these methods to the estimation of damage, factors such as uneven distribution of damage, opportunistic feeding habits and diets, and damage caused by different age classes should be considered. These factors are particularly important in horticulture, where fruit is often only a small proportion of a pest bird's diet (e.g. Adelaide rosellas (*Platycercus elegans adelaidae*) in cherry orchards: Reynolds, 2003).

In the case of bird damage, a number of factors make it difficult to estimate potential damage from observed bird numbers — required for any of the above -mentioned techniques. These include:

- unpredictability of bird movements, e.g. for species that do not maintain feeding pressure on a particular crop throughout ripening;
- difficulty in assessing bird numbers, particularly for small, mobile species or those that form large flocks when foraging;
- patchiness of bird feeding throughout a crop and between crops, and resultant spatially variable impacts;
- indirect impacts (for example, mould developing on pecked grape bunches); and
- compensatory production, so that the crop partly or wholly recovers from damage that occurs during development.

2.3. Control techniques

2.3.1 Bird scaring

Scaring and shooting are the most common approaches to pest bird control (Tracey, 2008). Birds are scared by unusual, sudden, unexpected, unfamiliar or dangerous events (scare stimulus), or by something that mimics a predator or the response to a predator (such as bird alarm calls). A bird's first reaction to being scared is flight. This is often followed by a period of curiosity, during which the bird tries to gather information about the scaring stimulus. Each time it encounters the stimulus, it gains more information. Eventually, it

accumulates enough information to know that unless the stimulus presents a real threat, it can be ignored — that is, the bird has become habituated to the stimulus (Thompson and Spencer, 1966). The time taken for habituation will vary, depending on a suite of factors, including species, surrounding habitat and the regularity and type of noise. Habituation is the single factor that most limits the effectiveness of scaring. It is also possible that once birds habituate to a stimulus, it could then work as a cue indicating the presence of available food (Conover and Perito, 1981). Under these circumstances it would attract birds to a crop and have the opposite effect to that desired.

Scaring is also likely to be more effective when alternative attractive feeding sites are available (Jarman, 1990; Crossfield, 2000). Most successful scaring of pest birds is achieved by using a variety of different scaring devices (Bishop et al., 2003), starting them as soon as birds show an interest in a crop and before the birds get into the habit of feeding there. Changing devices and moving them around frequently will also help to avoid habituation (Marsh et al., 1991).

Visual scaring methods

A wide variety of visual scarers are used. They include plastic shopping bags; car-yard bunting; spinning metal strips; reflective mirrors or tape; balloons displaying big eyes; and predator models such as scarecrows (human effigies), plastic silhouettes of birds of prey, or kites in the shape of predatory birds.

Balls or balloons with large eyespots are inexpensive scaring devices. Helium-or air-filled balloons with eyespots are tied to vegetation or to long poles. Tests of the effectiveness of eyespot balloons are mainly inconclusive (Marsh et al., 1991). McLennan et al., (1995) found that a commercial ball with a reflective eye that appeared to move as the ball rotated was significantly ($P < 0.01$) more effective at repelling sparrows from a feeding table than a beach ball with an eye painted on it, although the deterrent effect was minimal at 40 metres and ceased after nine days.

Kites shaped like birds of prey (often falcons or hawks) are another type of inexpensive visual scaring device. These are usually tethered to the ground, or may be suspended from helium-filled balloons that are tethered to a stake by a long monofilament line 30–60 metres above the ground. Ground-tethered kites require constant, low-velocity wind to

keep them aloft, but often in the early morning and late afternoon there is little wind and these are the times when some birds tend to feed. Winds of over eight kilometres per hour can blow down kites and balloons (Hothem and DeHaven, 1982), which may also be damaged when they become entangled in trellises or vegetation.

Predatory bird kites suspended from helium-filled balloons have successfully reduced bird damage to blueberries (damage reduced by 35%), vineyards (by 48%; range 32%–88%) and cornfields (by 83%) in North America (Conover, 1982; Hothem and DeHaven, 1982; Conover, 1984).

To be effective, the predator kites were used at a density of about one per hectare. The main cost when using the predator kites was maintaining the helium balloons, as most lasted only a few days. The kites were more effective against some pest bird species than others. The effectiveness of predator kites may be improved by selecting a model that closely resembles a predator species that occurs in the local area (Marsh et al., 1991).

Another inexpensive scaring device used is a predatory bird model mounted on a pole or building. For example, Conover (1985a) used animated owl models to protect vegetable plots from damage caused by American crows (*Corvus brachyrhynchos*). The owl model, grasping a crow model in its talons, was mounted on a weather vane so that it moved in the wind. The wings of the model also moved, either by the wind or by a battery-operated motor. This animated predator model reduced crop damage by 81% compared with an unprotected control plot and was relatively cheap to build.

In general, visual scarers offer only short-term protection, as birds quickly realise that they pose no real threat and then become habituated (Long et al., 1990; Marsh et al., 1991; McLennan et al., 1995). For example, some birds habituate to predator kites after only five hours' exposure (Conover, 1982). Visual scarers are simply something new and unusual in the birds' environment, and they soon learn to ignore them. This is particularly true for devices that are not kept in motion by wind or motor. Effectiveness also declines with distance from the scarer. For example, McLennan et al. (1995) found the effectiveness of eyespot balloons in keeping sparrows away from a feeding table was greatest at the closest distance measured (ten metres) and negligible at 40 metres.

Marsh et al. (1991, 1992) made some generalisations about scaring with scarecrow and predatory bird models on the basis of their review of the world literature on this topic. For best results, scarecrow and predatory bird models should:

- appear lifelike;
- have motion (for example, pop-up scarecrows and windblown predator models);
- be highly visible;
- be moved frequently to new locations in and around the crop to help prevent habituation;
- be supported by additional control methods, such as shooting to scare, or other acoustic scaring devices; and
- be started before birds develop a feeding habit in a crop.

Despite some old and resilient myths, birds do not seem to be scared by bird carcasses (Naef-Daenzer, 1983) unless they are life-like or in a threatening pose, and even then habituation develops rapidly (Bishop et al., 2003). Snake and cat models are equally ineffective (Marsh et al., 1991).

The major limitation of attempting to scare birds using reflectors, bright spinning or flapping objects, or similar devices is rapid habituation (Marsh et al., 1991). Wind conditions are important, because wind creates motion and sometimes sound, which increases the effectiveness of visual scaring devices (Tobin et al., 1988; Marsh et al., 1991). For example, CDs (compact discs) hanging on string in fruit trees cause random light flashes in the wind. However, high winds can break, or even blow away, scaring devices.

Once birds habituate they will fly between scaring devices or even perch on them before entering a crop to feed. Different pest bird species may have different responses to scaring devices. For example, in Ohio, red-winged blackbirds (*Agelaius phoeniceus*) and sparrows were effectively scared from grain and sunflower crops by reflecting tape, but American goldfinches (*Carduelis tristis*) and mourning doves (*Zenaida macroura*) were not (Dolbeer et al., 1986). Reflecting tape was also found to be ineffective for repelling starlings, American robins (*Turdus migratorius*), house finches (*Carpodacus mexicanus*), mockingbirds (*Mimus polyglottos*) and grey catbirds (*Dumetella carolinensis*) feeding in

blueberry plots (Tobin et al., 1988). Dolbeer et al. (1986) speculated that reflecting tape might be more effective against flock-feeding birds than those that feed solitarily or in small groups.

Acoustic scaring methods

Scaring with acoustic (sound-producing) devices, including ultrasonic devices, is often promoted as effective, scientific, humane, cheap and simple to operate (Bomford and O'Brien, 1990). The most commonly used acoustic devices rely on startling or fear for their scaring effects. Most are non-biological sounds generated by mechanical, electronic or explosive means and may include wind or mechanically powered noise generators, a range of electronically amplified sounds, propane gas cannons, crackers and firearms. Some devices produce bioacoustic sounds and others produce ultrasound (sound beyond human reception).

Sound travels through air in waves, and the loudness of sound, usually measured in decibels, declines with the square of the distance from the source. This means that the loudness of a signal drops away rapidly with distance. Sound shadows also form behind objects, such as trees or bushes, which further decrease sound signal strength (Marsh et al., 1991).

The most common form of scaring with sound relies on shooting to scare or harass, or devices such as gas guns. Shooting can reduce habituation (Baxter and Allan, 2008) and should be initiated before other scarers so that birds make a connection between the loud noise and danger.

Most electronic acoustic devices such as gas guns are set to go off automatically at either regular or random intervals; others are triggered by the movement of birds. As with visual methods of control, birds become habituated to, and then ignore, sounds after a time if they are repetitive, emanate from the same point source, or pose no physical threat. An extreme example is the lack of response shown by birds adjacent to airport runways as jet aircraft take off or land, often only metres away, with noise levels well over 100 decibels. A scaring and chemical repellent system that operates only when birds fly through a radio beam was shown to be more resistant to habituation than alternative systems that operated at regular or random intervals for keeping waterfowl away from contaminated ponds

(Stevens et al., 2000). There is at least one commercial device available in Australia that is triggered by radar detection of the birds and bioacoustic deterrent calls are activated by radio transmission (Muehlebach and Bracher, 1998). However, the effectiveness of such devices in crops has not been scientifically assessed.

Bioacoustic or biosonic sounds are broadcasts of recorded calls used in animal communication: usually alarm, distress or predator calls, or electronic mimics of such calls, are used in a variety of acoustic devices available on the market. The calls are recorded, sometimes digitised and modified, amplified, and then broadcast through speakers (Aubin, 1990; Marsh et al., 1991).

Some birds give alarm calls when they see a predator or something they perceive as a threat. Alarm calls alert nearby birds to the presence of danger, and the usual response is immediate flight. Alarm calls are often species-specific, although some species will respond to other species' alarm calls (Baxter et al., 1999). When taped alarm calls, or electronic imitations of alarm calls, are broadcast, they may have a similar effect to a real alarm call. The effectiveness of broadcast alarm calls for scaring birds away is likely determined mainly by the quality of the sound and by how often it is repeated.

Distress calls are usually loud 'squawks' given by birds held captive, either in a net or by a predator. The common response to a distress call is for surrounding birds to be attracted to the site, where they often fly around making a lot of noise in what is called mobbing behaviour (Conover and Perito, 1981; Conover, 1994a, b). Generally, distress calls are likely to be less effective for scaring birds than alarm calls, but distress calls are sometimes used because they are easier to record, and they have been shown to be effective for dispersing herons (Ardeidae), gulls (*Larus spp.*) and crows (Naef-Daenzer, 1983; Gorenzel and Salmon, 1993; Bishop et al., 2003).

Birds habituate rapidly and start to ignore a broadcast alarm or distress calls if the same call or call sequence is frequently repeated (Aubin, 1990; Martin and Bateson, 1993; Yokoyama and Nakamura, 1993; Harris and Davis, 1998). They may take flight, but rapidly return to continue feeding. The quality of the broadcast sound is determined by the quality of the recording and the quality of the amplifier and speakers used to broadcast the call. The broadcast calls generally need to be those of the bird species present, or at least

calls from species the local birds usually respond to. Birds have dialects, and the alarm call of a bird from an area with a different dialect may be less effective than a locally recorded call (Marsh et al., 1991).

The calls of birds of prey or imitations are sometimes used to scare birds. There is little evidence in peer-reviewed literature that such sounds are effective. In fact, many predators do not call when they hunt, as it would make little sense for them to call out and warn potential prey. In cage tests, Yokoyama and Nakamura (1993) found that for young tree sparrows (*Passer montanus*), the sound produced by a paper flag was significantly ($P < 0.05$) more aversive than a broadcast distress call of their own species. The distress calls were also subject to rapid habituation.

Bird vocalisations, including alarm and distress calls, are extremely intricate. Birds are more likely to accurately interpret pre-recorded amplified sounds when high-quality recording, amplifying and broadcasting equipment is used (Aubin, 1990; Marsh et al., 1991). In addition, if calls are recorded, digitised, stored on a computer chip and then amplified through speakers, there may be a marked reduction in the aversive stimuli contained in the calls. It is possible that such bioacoustic sounds represent little more than something new and unusual in the birds' environment, and different devices simply present different sounds.

It is possible that 'communication jamming' occurs when sounds with a similar frequency range to birds' communication calls are broadcast (Rooke, 1983). This supposedly inhibits some flock-feeding birds such as silvereyes from hearing each other so they become confused. Some devices are designed to produce sounds that irritate, rather than scare or distress, to limit habituation. However, this has not been investigated.

Ultrasound is very high frequency sound above the range of human hearing (greater than or equal to 20 kilohertz). Most bird species cannot hear ultrasound, or they can hear only the lower frequencies (Beuter and Weiss, 1986; Marsh et al., 1991). Even for birds that can hear ultrasound, there is no reason for it to be more effective for scaring than audible sound. Despite anecdotal user testimonials and unsubstantiated claims from advertisers, manufacturers and distributors, no scientific field experiments have indicated that ultrasound is of value for reducing bird damage to crops. In fact, experiments have shown

that ultrasonic devices are ineffective (Bomford, 1990a; Bomford and O'Brien, 1990; Erickson et al., 1992; Haag-Wackernagel, 2000).

Few reliable scientific experiments have been conducted on the value of acoustic devices for reducing bird damage to crops. However, on the basis of reviews of the world literature on this topic, Bomford and O'Brien (1990) and Bishop et al., (2003) drew some generalisations about scaring with sound.

They suggest that the best effect is obtained when:

- the sound is presented at random intervals;
- a range of different sounds is used;
- sounds are broadcast for the minimum time needed to get a response;
- the sound source is moved frequently;
- the sound is supported by other control methods; and
- the sound is reinforced by real danger, for example, shooting.

Bomford and O'Brien (1990) and Bishop et al., (2003) also suggest that:

- loud sounds are more aversive than quiet sounds (if the frequencies are within the birds' hearing range);
- sounds with a wide frequency range are more aversive than pure tones;
- loud sounds produced by simple, inexpensive methods can be as effective as sounds produced by expensive devices;
- adult birds are more easily scared than juveniles;
- hunted species take longer to habituate to bangs;
- broadcast alarm and distress calls can be effective but are subject to habituation and are often species-specific; and
- all species eventually habituate to nearly all sounds tested.

Combining visual and acoustic scaring methods

The best results are likely to be obtained if different control methods are combined to prevent habituation (Bishop et al., 2003). For example, distress calls can attract birds (Conover, 1994a) and starlings and American crows habituated less to plastic owl models when they appeared to be grasping a struggling bird (Conover and Perito, 1981; Conover, 1985a). Nakamura (1997) found that playbacks of taped calls of jungle crows (*Corvus*

macrorhynchos) were largely ineffective for scaring rufous turtle doves (*Streptopelia orientalis*), as was the presentation of a stuffed jungle crow. However, the combined stimuli of a stuffed crow with a crow call were highly effective and were resilient to habituation in the three successive trials conducted.

Cummings et al., (1986) evaluated a mechanical, gas-operated, pop-up, life-size, human scarecrow model coupled with a propane exploder for reducing red-winged blackbird damage to sunflower crops. The device was set so that the exploder went off 15 to 30 seconds after the scarecrow popped up (Cummings et al., 1986). In three fields, damage was reduced by an average of 84% in the first five-day treatment period and by 59% in a subsequent five-day treatment period. In two other fields near roost sites where red-winged blackbirds were well established, damage was reduced by only 8% and 31%. Cummings et al., (1986) concluded that the scarecrow-exploder device would be economically worthwhile for crops in which damage levels exceeded 18%, which was about 1.2% of crops.

Scaring with aircraft

The use of model aircraft, ultralights or full-sized aircraft to chase birds from crops is an example of combining visual and auditory stimuli. Garrity and Pearce (1973) found that model airplanes controlled by skilled operators reduced the numbers of robins in blueberry fields, but they also achieved only partial coverage of the crop at risk. The robins resumed feeding during refuelling and soon after flights ceased.

Birds of prey

Falconry or ways of attracting true predatory species to remain close to a crop is often the subject of inquiry. Trained falcons and hawks are sometimes used to keep birds away from overseas airports (Erickson et al., 1990). However, the efficacy of this has rarely been evaluated. In one study, trained falcons used at O'Hare International Airport to keep birds away from the runways were not as cost effective as two men with shotguns (Dolbeer, 2003). Previous attempts to use falconry to protect agricultural crops have mostly been unsuccessful.

In North America, artificial perches and nest boxes were provided in orchards in an unsuccessful attempt to attract birds of prey to reduce vole populations (Askham, 1990).

To attract birds of prey to vineyards, Howard et al., (1985) hung live decoy prey birds in cages from artificial perches. Hawks attacked the caged birds, but nearby feeding birds simply moved a short distance to other sections of the vineyards and grape damage was not reduced. Perches placed around the perimeter of irrigated soybean crops to enhance house mouse predation significantly ($P < 0.001$) increased the number of diurnal raptors visiting and hunting over these crops, compared with untreated crops (Kay et al., 1994).

Encouraging raptors to specific areas is problematic, as different species occupy different ecological niches. For example, sparrowhawks and goshawks (*Accipiter spp.*) prefer hunting amongst trees and tall shrubs to surprise prey. Conversely, most falcons prefer open country, and Australian hobbies prefer lightly timbered country along watercourses (Marchant and Higgins, 1993). The most effective predators of adult pest birds are also unlikely to be attracted by carrion or other food sources. Species most likely to be attracted to carrion (such as wedge-tailed eagles, *Aquila audax*, little eagles, *Hieraaetus morphnoides,* and whistling kites, *Milvus sphenurus,* do not normally hunt birds in flight. Some studies have shown that providing perches increases the numbers of birds of prey (Kay et al., 1994). However, this has not yet been demonstrated to reduce the number of pest birds or the damage they cause.

2.3.2 Population reduction

Most attempts to reduce damage by reducing pest bird populations have been in-effective (Dyer and Ward, 1977; Feare et al., 1988; Feare, 1991). To attempt long-term population control for any species, a good understanding of their population dynamics (Murton, 1968; Dolbeer, 1998) and the subsequent effects on the environment is essential. In many circumstances it is not uncommon for up to 65% of young birds born each year to die before they are one year old (Feare, 1984).

Lethal control is often ineffective for species with high reproductive capacity and high rates of annual mortality. For example in Belgium, seven years of substantial effort using explosives resulted in almost 750,000 starlings being killed at their roosts in an attempt to reduce damage to cherry orchards (Tahon, 1980). However, because of high immigration and recruitment rates this had little medium- or long-term influence on starling populations or crop damage.

For pest birds with high reproductive rates, control during breeding may be more effective than control at other times of the year (Paton et al., 2005). This may be the case for starlings, because large numbers of juveniles congregate after breeding (Feare, 1984), which coincides with the grape-ripening season. However, for birds with low reproductive potential, lethal control can be up to six times more efficient than reproductive control (Dolbeer, 1998). Hence population dynamics and targeting of the timing of control are important considerations.

Feare (1991) suggests that there are two fundamental reasons why attempts to reduce pest bird populations over broad areas have failed. First, most pests have a wide geographical range and much of the population is inaccessible to control operations. Second, control attempts can be counteracted by compensatory increases in breeding and survival. There are many examples where population control has not been successful for these reasons. Some examples are the aerial application of organophosphate for controlling quelea (*Quelea quelea*) in Africa (Ward, 1979); shooting wood-pigeons (*Columba palumbus*) to reduce damage to grain and clover in Cambridgeshire, United Kingdom (Murton et al., 1974); application of the surfactant PA-14 to large roosts of common grackles (*Quiscalus quiscula*), red-winged blackbirds and starlings to reduce agricultural damage in Tennessee, North America (White et al., 1985); and the use of explosives to control starlings at roosts to reduce damage to cherries in Belgium (Tahon, 1980). However, when dealing with a small, isolated population where immigration is preventable, a substantial reduction in numbers may be achievable (Feare 1991; Dolbeer 1998).

Short-term population reduction, such as concentrated efforts in small areas during critical ripening periods just before crop damage occurs, may be effective. Ward (1979) proposed that an ‘immediate crop protection’ strategy for quelea around cereal crops would be preferable than the previous ‘total reduction strategy’. This was proposed after an estimated one billion quelea were killed annually by aerial spraying with avicides, with no indication of long-term reductions in population levels or damage. No published evidence could be found showing either short-term or long-term population reduction leading to reduced crop damage.

While shooting is commonly used (Fleming et al., 1990b), few studies have evaluated its efficacy. For example, studies of wood pigeons and damage to brassica crops showed that

an intensive shooting campaign in the experimental area did not result in less damage than at control sites (Murton and Jones, 1973). Further, studies showed that shooting did not increase the winter mortality of wood pigeons above the level experienced in the absence of shooting Murton 1974 (Murton et al., 1974).

Many devices have been used over the centuries to trap or snare live birds (MacPherson, 1897; McClure, 1984; Bub, 1995). Traps have allowed the capture of birds for many purposes including harvesting (Dei, 1989), control and management of localised (Conover and Dolbeer, 2007; Tracey et al., 2008b) or establishing (Campbell et al., 2012) pest populations, research, banding and monitoring (Lowe, 1989; Dieter et al., 2009; Krementz et al., 2011) and for disease surveillance (Tracey, 2010).

On the basis of a nationwide questionnaire conducted in the USA, Gorenzel et al., (2000) reviewed trapping for pest bird control. Most respondents (57%) considered that trapping was not important overall for bird control in crops. However, in California, some respondents thought trapping was important for control of starlings and house finches in grapes. Gorenzel et al., (2000) found no rigorous evaluations of the effectiveness of trapping or the factors influencing results. Most evaluations of trapping put an emphasis on the numbers of birds caught rather than on damage levels in relation to the cost of control.
The most common trapping mistakes listed by respondents were failure to conduct adequate free-feeding and poor trap placement (Gorenzel et al., 2000). Free-feeding (also called pre-baiting or pre-feeding) is where bait is placed out for several days before traps are activated. Poor trap placement was probably due to inadequate observations of flight paths and roosting and feeding areas. Failure to use decoy or call birds in traps can also influence their effectiveness (Williams and Schwab, 1974).

A trap that has been used with some success on a variety of bird species is the modified Australian crow (MAC) trap (Elliott, 1964; Larsen and Mott, 1970; Moran, 1991; Moran et al., 2004; Conover and Dolbeer, 2007). This design was first developed to capture crows (Woodbury, 1961). The V-shaped entrances of this trap can be adjusted for different species (Gadd, 1996).

A variety of single-catch nest box traps have been used to capture hole-nesting species (DeHaven and Guarino, 1969; Stewart, 1971; Blums et al., 2000). Dehaven and Guarino

(1969) used a spring-loaded trap door that closed over the entrance of the nest box when triggered by a treadle inside the box. More sophisticated designs use electronics for monitoring captures (Stewart 1971). Stewart (1973) operated a single nest box trap during a 124-day period and captured 56 starlings. Knittle and Guarino (1976) used 26 nest box traps in approximately 80 hectares (200 acres) and captured 294 starlings in 57 days. On the basis of the reproductive capacity for the area (Dehaven and Guarino, 1970), they concluded that this achieved an overall reduction of about 959 birds from the post-breeding population (Knittle and Guarino, 1976). They also suggested that a multi-catch design would greatly improve the efficiency of catching starlings and may be of benefit for small fruit orchards.

Poisoning

The main limitations with avicides are illegal poisoning (Du Guesclin et al., 1983); community resistance to their use; their impacts on non-target species; animal welfare concerns; and (depending on the poison used) their residual or secondary effects in the food chain. Ensuring adequate free-feeding is the most effective way to reduce bait-shyness (Nelson, 1994) and limit non-target effects.

A range of poisons has been used to kill birds. They include organophosphates (Ridpath et al., 1961), endrin (Stickel et al., 1979), 1080 (Balcomb et al., 1983), nicotine (Ridpath et al., 1961), strychnine (Long and Vagg, 1960; Ochs, 1976; Redig et al., 1982), PA-14 (Heisterberg et al., 1987) and brodifacoum (Godfrey, 1986; Porter, 1996). Poisons currently registered for pest bird control overseas (DRC-1339) or in Australia (4-aminopyridine, alpha-chloralose and fenthion) are discussed in more detail below.

Most of the avicides discussed in this section have not been demonstrated to effectively reduce damage caused by pest birds. There are also animal welfare and target specificity concerns associated with many avicides, and there has been little assessment of their potential non-target effects on Australian species.

<u>DRC-1339</u>

DRC-1339 (3-chloro-4-methylaniline hydro chloride, Flockoff® or Starlicide®) is a poison that affects renal function in birds. It is currently not registered for use in Australia. This poison was identified by the Denver Research Centre (DRC) after evaluating more

than 2000 chemicals for pest bird control between the 1940s and the 1980s (Spurr, 2002). In North America and New Zealand it has been used for over 30 years (Bull, 1965; Besser et al., 1967), and it is currently applied to cereals, cereal pellets, bread and dripping, and sultanas for controlling starlings, red-winged blackbirds, crows, ravens and gulls.

In New Zealand it has been used for many years to control rooks. Initially, ground baiting of rooks was conducted using bread and dripping at carefully selected times of the year when their preferred foods were lacking. These control operations were very successful in terms of numbers of birds killed (over 86,000 were killed in the first 15 years) (Porter, 1987). More recently, a jellied form of DRC-1339 has been applied to the edges of nests by an operator hanging from a helicopter (Porter et al., 2008).

DRC-1339 is unique among avicides, as it has selective toxicity for different bird species. Many species that are regarded as pests, including starlings, pigeons, gulls, crows and ravens, are highly sensitive (United States Department of Agriculture, 2001; Eisemann et al., 2003). Conversely, DRC-1339 has been shown to have low toxicity to most mammals (except cats) and many bird species native to North America (Eisemann et al., 2003). Of the 55 bird species tested, two are native to Australia: the budgerigar (*Melopsittacus undulatus*) and the barn owl (*Tyto alba*). As this toxin is metabolised rapidly there is minimal risk of secondary exposure. The mode of action is irreversible kidney and heart damage, which causes death three to 50 hours after ingestion (United States Environmental Protection Agency 1995 (United States Environmental Protection Agency, 1995).

Aminopyridine

Aminopyridine (also called 4-aminopyridine or Scatterbird®) has effects similar to those of central nervous system stimulants. When birds consume treated grain, it causes them to behave erratically and to give off distress calls before death (Goodhue and Baumgartner, 1965; Gadd, 1992). This may frighten away nearby birds or cause them to mob the affected bird. Hence this chemical is also considered a secondary chemical repellent. There are animal welfare, social perception, target specificity and human safety concerns about the use of this chemical, and it is unlikely to gain registration in other States and Territories.

Alpha-chloralose

Alpha-chloralose (or α-chloralose) is a chloral derivative of glucose that acts as a soporific or narcotic by depressing the cortical centres of the brain. As a soporific it is the most humane of the avicides. Alpha-chloralose can be mixed with grain bait at a concentration of around 2% and offered to birds after a period of free-feeding (Nelson 1994). Alternatively it can be added to drinking water. Care needs to be exercised with the use of alpha-chloralose to avoid bait shyness. An advantage of alpha-chloralose, particularly where non-target species may be at risk, is that the dose can be reduced so that birds are immobilised and not killed. Non-target species can be revived and released and target birds can be killed humanely. The dose rate that causes mortality varies with the species, the size of the bird, and the ambient temperature. Higher mortality is evident in smaller birds and at low (< 12 °C) or high (> 30 °C) air temperatures.

The main use of alpha-chloralose is for controlling feral pigeons around buildings. It has also been useful for removing small or establishing populations of sparrows, starlings, mynas and crows. When used for bird control, alpha-chloralose usually kills few individuals but causes the bulk of the population to disperse, and this may last long enough for a crop to ripen. Hence, it may be considered to act more as a chemical repellent than as a poison. However, non-target species may be at risk. Sinclair and Cerchez (1992) trialled alpha-chloralose on sparrows in apricot and grape crops and on starlings in a cattle feedlot. With sparrows, they found that 0.5% weight/weight alpha-chloralose on mixed canary seed resulted in variable mortality rates between trials. However, the baiting caused dispersal of local flocks away from the crops. The dispersal lasted up to 90 days, which was long enough for the crops to be harvested (Sinclair and Cerchez, 1992). At the cattle feedlot, starlings were successfully dispersed using 1.5% weight/volume alpha-chloralose in water or 1.5% weight/weight alpha-chloralose in cattle feed placed outside but adjacent to the feedlot troughs where the birds were foraging on split food.

Fenthion methyl

Fenthion methyl (commercial names include Control-a-Bird®, Rid-a-Bird®, Avigel® and Avigrease®) is an organophosphate that acts as a cholinesterase inhibitor and neurotoxin. The chemical is usually mixed in a special grease or gel for surface application inside buildings and on structures such as bridges and steel girders. Birds get the grease on their feet and the poison is absorbed through the skin. The chemical is rapidly metabolised in birds, thus reducing the risk of secondary poisoning. Non-target species may succumb to

primary poisoning if they consume the feet or beaks of birds poisoned by the grease (Hunt et al., 1991, 1992). This chemical is not available as an oral toxin, and its use has non-target (Bruggers et al., 1989), welfare (Spurr, 2002) and human health (Jeremiah and Parker, 1985) concerns. Because of these issues, it is unsuitable for protecting horticultural crops.

Chemical fertility control

A number of chemical products cause infertility in birds when added to their food. Reproduction is also prevented when chemicals or oils are sprayed on their eggs. No published evidence could be found demonstrating that fertility control chemicals can reduce pest bird damage to crops. A drawback of many fertility control agents is that they require several doses. There is little information about the effects of these products on offspring that do hatch but may have received a partial dose. Oestrogen-based products are likely to affect the fertility and sexual development of any non-target species taking bait.

Bomford (1990b) reviewed chemical fertility control techniques and assessed the potential value of several chemicals that reduce fertility in birds, including the following:

Mestranol

Mestranol (17-ethynyl-3-methyl ether) is an orally active oestrogen. In a cage trial, spraying the eggs of Japanese quail (*Coturnix coturnix*) with mestranol increased embryo and chick mortality, and made all quail that hatched irreversibly sterile (Wentworth et al., 1968). Force-feeding mestranol-impregnated grit to adult quail reduced their fertility, but this was not developed as a technique suitable for use on wild birds (Wentworth et al., 1968).

BDH 10131

BDH 10131 (the 3-cyclopentyl ether of 17αhexa-1′,3′diynyloestra-1,3,5(10)-trien-17β-ol) is a synthetic oestrogen that was investigated as an alternative to mestranol or quinoestrol because it was shown to be active for a longer period in laboratory rats (Kendle et al., 1973). In laboratory trials on birds, Kendle et al., (1973) fed BDH 10131 to caged pigeons (*Columba livia*) for two days and found that fertile egg production dropped to less than 20% of that in untreated birds.

Ornitrol®

Ornitrol® (20, 25-diazocholesterol hydrochloride) is a steroid that is a long-acting inhibitor of ovulation in many bird species and also inhibits testicular growth. When added to food at 0.1% it has been shown to delay or reduce egg production in caged and wild pigeons for up to six months (Elder, 1964; Wofford and Elder, 1967; Woulfe, 1968). At this concentration it took seven days for birds to ingest an adequate dose, but at a 1% concentration the birds refused to consume enough grain to be effective. Higher doses can also be toxic to birds (Lofts et al., 1968), and the signs described could have welfare implications. Wofford and Elder (1967) concluded that two treatments a year at 0.1% would control fertility if both treatments were timed to coincide with the breeding season.

Ornitrol® at 0.1% or 0.05% on grain fed to field populations of red-winged blackbirds had variable success, reducing hatch success by between 7% and 61% in various trials (Fringer and Granett, 1970). Timing of baiting, variable uptake of bait and promiscuity were proposed as factors reducing success rates. Canary seed impregnated with Ornitrol® at 0.1% and fed to captive sparrows resulted in 0% hatch success compared with 64% in a control group (Mitchell et al., 1979). This effect is not permanent, as a fertile egg was produced about a week after treatment ceased. Within a month hatch success was similar to that in the control group.

Triethylenemelamine

Triethylenemelamine (TEM) (2,4,6-tris(ethyle– nimino)-s-triazine) arrests spermatogenesis through inhibition of meiosis. Vandenbergh and Davis (1962) field-tested TEM on a breeding population of red-winged blackbirds in a marsh for two years. In both years the hatch rate was significantly reduced relative to that at a control site. In contrast, Fringer and Granett (1970) and Guarino and Schafer (1974) field-tested TEM on territorial male red-winged blackbirds and found that it did not reduce breeding success. Davis (1961) found that caged starlings orally dosed with TEM in winter, when the testes were fully regressed, did not recover their fertility for several months. A small field trial of TEM on male starlings, which were captured, dosed with TEM and released, showed that their breeding success was reduced. However, the production of some fertile eggs in the territories of sterilised males indicated that their female partners were occasionally mating with other males.

ThioTEPA

ThioTEPA is the abbreviation for triethyleneth iophosphoramide (tris (1-aziridinyl) phosphine sulfide). Potvin et al. (1982a, b) sterilised wild male red-winged blackbirds by feeding them thioTEPA-treated corn for ten days. The hatch rate was 46%, which was significantly lower than the average hatch rate of 85% in the control area. The fertility of some female partners of treated male red-winged blackbirds was suggested to have been a result of females copulating with males from other territories.

Nicarbazin

Nicarbazin (CH N O), is a complex of two compounds, 4,4'-dinitrocarbanilide (DNC) and 4,6-dimethyl-2-pyrimidinol (HDP). Nicarbazin is an oral contraceptive for birds and is registered by the United States Environmental Protection Agency for use against pest geese and pigeons. DNC is the active component but it is very poorly absorbed and requires HDP for absorption and to achieve a contraceptive blood level. Once absorbed, nicarbazin interferes with the formation of the vitelline membrane, separating the egg yolk and egg white. The effect on hatchability is a function of time and dose and is reversible. Nicarbazin must be consumed daily, consistently and in adequate quantity to achieve a contraceptive effect and a single or intermittent dose will not affect egg hatchability.

Smaller birds, including passerines, have the most inefficient absorption of nicarbazin, requiring a higher bait concentration and dose (Avery et al., 2008). A pigeon requires a dose rate of 83 mg/kg bodyweight/day. Assuming similar values for passerines, a 150 gram passerine would need to consume 5 g of 0.25% nicarbazin bait/day to obtain the recommended dose for effective contraception. Many passerines might require even higher doses. Delivering such high, consistent daily doses throughout the breeding season would be difficult for most birds that damage horticulture.

Egg oils

Vegetable and mineral oils can be used to prevent hatching when the oils are applied directly to eggs in the nest. An advantage of applying oils, rather than destroying eggs or nests, is that birds may continue incubating, in some cases beyond the normal time for hatching (Christens and Blokpoel 1991; Cummings et al., 1997). For many bird species re-nesting is common after nests and eggs are destroyed. Vegetable and mineral oils prevent the hatching of 96% to 100% of the eggs of chickens (*Gallus gallus*), ring-billed gulls

(*Larus delawarensis*), herring gulls (*Larus argentatus*) and Canada geese (*Branta canadensis*) (Blokpoel and Hamilton, 1989; Christens and Blokpoel, 1991; Baker et al., 1993; Christens et al., 1995; Cummings et al., 1997; Pochop et al., 1998a; Pochop et al., 1998b).

A study comparing mineral oil with commercially available oils (including castor, corn, linseed, safflower and soybean) found that they were equally effective (Pochop et al., 1998a). Preventing the hatching of eggs by using oils is effective, but may have a high labour cost due to the inaccessibility of many bird nests. Therefore this technique may only be useful for small or isolated pest populations (Miller, 2002). There may be an application for reducing small urban populations of pest birds with extended breeding seasons, such as ibis (Threskiornithidae) (Martin and Dawes, 2005).

2.3.3 Habitat management and decoy feeding

With increasing regulatory and social restrictions on killing birds or using noisy scaring devices, there is greater interest in manipulating habitat quality as an alternative means of reducing bird damage (Van Vuren, 1998). A number of approaches (reviewed by Bishop et al., 2003) can be applied. Habitat quality can be reduced so that fewer resources are available for a pest species and their numbers decline, or the crop can be made less attractive to pests. Alternatively, pest birds can be lured away from an area by providing more attractive habitats or food elsewhere.

Reducing habitat quality

St John (1991) found that modifying access to food and water reduced the number of little corellas roosting in river red gums (*Eucalyptus camaldulensis*), and alleviated damage. Experimental trimming of roost trees in Houston, Texas to reduce the urban impacts of brown-headed cowbirds (*Molothrus ater*), starlings, grackles (*Quiscalus quiscula* and *Cassidix mexicanus*), red-winged blackbirds and American robins was effective in preventing roosting (Good and Johnson, 1976, 1978). Trimming consisted of removing one-third of the canopy; this is considered a ‘heavy’ trim by professional tree surgeons. Stands of pruned trees were not occupied, whereas trees that were not pruned were occupied to the same level as in previous seasons (Good and Johnson 1976, 1978). Removing nearby food sources may also reduce damage on a local scale. For example,

removing blackberry bushes that are exploited by starlings, rosellas and silvereyes may help reduce damage to nearby fruit.

Some bird problems can be reduced by decreasing the attractiveness of sites. For horticulture, the varieties of fruit grown can be important with respect to both time of maturity, sugar content and type, fruit size, colour and texture. Depending on the main species of pest birds in an area, some varieties of fruit may be less prone to damage. For example, the fruit of some olive varieties may be too small or too large to suffer high levels of bird damage from particular species (Mladovan, 1998; Spennemann and Allen, 2000). Growers may be able to avoid growing varieties most prone to damage from information obtained from established growers in an area.

There is a range of factors that influence the severity of bird impacts. These factors may provide opportunities for reducing bird problems. For example, the only crop in an area or the first (Baker, 1980a,b) or last crop in a district to have fruit maturing are more likely to sustain bird damage. Orchard location can be important. For example, proximity of the orchard to either native vegetation, windbreaks consisting of exotic species, or powerlines may increase fruit losses caused by some species (Stevenson and Virgo, 1971; Graham, 1996). Land use around an orchard will also be important, as it will influence the availability of alternative foods. For example, there may be an association between livestock and starlings, as these birds regularly feed on ground-dwelling insects and grazing makes these insects more accessible to the birds. Isolated orchards tend to suffer more damage than those surrounded by other orchards producing similar fruit.

The pasture sward in an orchard and the surrounding area may influence damage levels. When planted in an adjacent field it may offer an alternative (decoy) food that helps to attract the birds away from the fruit. Conversely, pasture within orchard rows may provide food that attracts birds, and when the crop ripens it becomes an additional food item for birds. For example, in New Zealand, orchards that have a sward of grass that seeds in late winter or early spring attract birds such as sparrows and greenfinches (*Carduelis chloris*), and these species will nip the fruiting buds of apples and pears, causing losses as high as 90% (Richard Porter, Havelock North, New Zealand, pers. comm. 2005). Regular management and maintenance of pasture swards between rows can reduce the alternative food for some pest species.

Depending on the grass species and height, the pasture sward can either increase or decrease the abundance or availability of certain insects, and this in turn may influence bird damage in different ways. For example, starlings prefer short (Whitehead et al., 1995) and freshly mown (Tinbergen, 1981) grass where insects are more accessible. Woronecki et al. (1981) and Woronecki and Dolbeer (1980) found a strong and consistent relationship between reduced insect populations and reduced corn damage by red-winged blackbirds in Ohio. Conversely, in New York, reduced damage by the same bird species in corn was found to be related to increases in insect populations (Bollinger and Caslick, 1985b).

It is important to observe the birds responsible for crop damage and their behaviour patterns. Forde (1989) recommended planting rows of alternative food such as sudax grass to reduce damage to fruit by regent parrots (*Polytelis anthopeplus*) and yellow rosellas (*Platycercus elegans flaveolus*) because he observed that the birds preferred sudax seed to other native seeds, commercial seeds, fruit or nuts. Reynolds (2003) observed Adelaide rosellas foraging on soursob bulbs (*Oxalis pescaprae*) in cherry orchards and showed that the birds could be attracted to feed plots where the bulbs had been made available by light cultivation. He concluded that a number of weed or pasture species could be similarly manipulated to act as decoy foods but suggested that measures encouraging birds to feed elsewhere should be counter-balanced by an integrated approach, discouraging them from feeding in the susceptible crop.

Decoy food

Growing decoy crops has been successfully used to reduce bird damage to sunflower crops (Broome, 1979; Allen, 1982, 1984; Cummings et al., 1987). Providing alternative food sources for horticulture requires careful consideration of the pest species and their preferences and feeding behaviour.

A decoy crop needs to be at a stage of maturity where birds will feed on it just before the grower's commercial crop becomes vulnerable to attack, so that the birds' feeding patterns are established on the decoy food. Scaring can be used in conjunction with decoy feeding and should be concentrated around the orchard and kept well away from the decoy site. It may take more than one season to develop established feeding patterns on a particular decoy site. If decoy food (rather than a decoy crop) is supplied, it must be highly palatable

and at least as nutritious as the commercial crop, otherwise there is little reason for birds to be attracted to it.

The strong attraction starlings have for soil insects may offer an opportunity to exploit a particular feature of a pest species' diet by using a 'decoy feeding' strategy. When fruit matures in late summer or autumn, soil insects are often largely inaccessible to starlings because the soil is dry and hard. Keeping an area of ground moist may improve access to soil insects, the preferred food source. However, this strategy has not yet been proven to reduce fruit losses, and omnivorous birds may still consume fruit preferentially when it is available.

Native vegetation

Native flowering plants can be planted to act as decoy food sources for native honeyeaters. Increasing plant diversity and the extent of native vegetation on farmland is known to increase the diversity of birds, particularly native species (Green, 1986; MacDonald and Johnson, 1995). This leads to the perception that damage to fruit crops will be amplified with increased plantings of native vegetation. However, many pest birds, including crows, ravens, starlings, cockatoos and corellas, prefer open agricultural areas. Other species, such as blackbirds (*Turdus merula*) and mynas, thrive in urban environments. Pied currawongs (*Strepera graculina*) and noisy miners (*Manorina melanocephala*) thrive in fragmented habitats with little structural diversity. Increasing the extent of well-structured and diverse native vegetation may not increase the abundance of these species. The bird species, the plant species and their times of flowering, and the structure and extent of vegetation will determine whether plantings serve as decoy food sources or whether they attract more damaging species.

Many birds, including honeyeaters and silvereyes, are attracted to nectar-producing trees and shrubs. They will preferentially feed on these plants rather than on fruit crops. When the surrounding vegetation produces good quality nectar, bird damage is often low. When investigating nectar flows in the Margaret River, Rooke (1983) found that higher average yield of honey per hive corresponded with lower damage by silvereyes to grapes. Bird damage was lowest during good nectar years, which coincided with warm springs and autumns and relatively cool periods during February and March. Further research suggested that silvereyes prefer alternatives to grapes, including sugar-water and plants

such as marri, seaberry saltbush (*Rhagodia candolleana*), nightshade (*Solanum spp*.), berries and figs. Research has also found that birds damaging grapes were usually in poor physical condition, possibly because of a lack of natural food sources. Rooke (1983) also discovered that providing additional food did not increase the number of silvereyes.

Native flowering plants also attract insectivores, including many honeyeaters. These bird species may be beneficial in the vineyard throughout the year by controlling insect pests. Providing well-structured native vegetation can serve to provide shelter for insectivores, support bird diversity, and supply an effective decoy food source. Selecting the most appropriate plant species is crucial to ensure that the nectar source is acting as a diversion from the orchard rather than attracting more pest birds. Habitats with exotic flowering plants can be preferred by introduced bird species such as starlings and blackbirds (*Turdus merula*) (Green, 1986; Williams and Karl, 1996; Kinross, 2000) and native frugivores (Recher and Lim, 1990) that damage fruit. Many birds beneficial in vineyards are absent from introduced vegetation such as pines. The absent species include specialist predators, Eucalyptus canopy feeders, obligate cavity-nesters and insectivores (Suckling et al., 1976). These species can control harmful insects or compete with, or prey on, pest birds.

A balance of native shrubs and trees of varying heights is recommended for conservation and may reduce the numbers of pest birds. To avoid colonisation by aggressive edge-specialist honeyeaters (for example, noisy miners), O'Neill (1999) suggests that revegetation should not include more than 20% of nectar-producing shrubs. Providing excess nectar in winter may also cause normally non-sedentary species, such as silvereyes or lorikeets, to overwinter in orchards. Plantings of marri, figs, banksia and seaberry saltbush are recommended to reduce silvereye damage to grapes in the south-west of Western Australia (Rooke 1983).

An awareness of the main bird species in an area is important in deciding the most suitable plant species and where to plant them. Plantings should be located where they are most likely to attract birds and far enough away from the orchard to avoid damage. Ideal decoys for honeyeaters will be those plants that flower before a commercial crop becomes vulnerable to attack and that continue to produce nectar throughout the ripening period.

The flowering periods of decoy plantings and how this relates to the ripening times of the varieties present on the property need to be considered. Abundant nectar just before or after ripening can inadvertently result in increased damage. For example, large numbers of noisy friarbirds damaging vineyards in Orange, New South Wales, have been linked to heavy flowering of red stringybark (*Eucalyptus macrorhyncha*) in the same season (Tracey and Saunders, 2003). In that season, harvesting was delayed by adverse weather. Noisy friarbirds attracted to flowering red stringybark in the area, then switched to feeding on mature wine grapes after nectar loads were exhausted. Hence it is important to select decoy trees and shrubs that are productive for the whole period that crops are vulnerable to bird damage. For honeyeaters, the preferred species for decoy plantings include *Eucalyptus spp., Melaleuca spp., Callistemon spp., Banksia spp.* and *Grevillea spp.*. These plants may attract insectivores and serve as decoy food sources for native honeyeaters and silvereyes.
In summary, the most appropriate plants to act as a decoy food source will depend upon; the pest bird species; the time of ripening for the varieties grown; climate; and soil type.

Locally indigenous plant species are less likely to become weed problems and are more likely to be attractive to local bird species. The use of decoy plantings can be risky because of seasonal variations in the timing of flowering. This control technique should be used with caution and in conjunction with other control methods.

2.3.4 Exclusion

Netting

Exclusion netting is widely used and is an effective way of reducing or preventing damage (Stucky, 1974; Foster, 1979; Fuller-Perrine and Tobin, 1993; Tracey and Vere, 2007). As bird damage is often variable and difficult to predict, one of the attractive features of exclusion netting is that it reduces uncertainty and the need to monitor the bird problem. Netting also overcomes increasing concerns about the use of chemicals, animal welfare issues, and restrictions on the use of acoustic devices under noise pollution control legislation.

Netting is not the best solution in all situations. It is an acceptable solution when the benefits from excluding birds and not having to carry out any other bird control exceed the costs of netting. Permanent netting is unlikely to be an economic solution for low-value

crops or for crops that usually sustain only a low level of bird damage (Hector, 1989; Sinclair, 1990; Slack and Reilly, 1994).

Drape-over or throw-over nets, although previously used mainly in home gardens and on small hobby farm tree crops, are now becoming increasingly common on commercial horticulture crops — particularly high-value grape and berry crops. They offer short-term protection over the ripening season. Drape-over nets are lightweight, relatively inexpensive, extruded or loosely knitted fabrics that are available in a variety of colours, mesh sizes and widths (Duffy, 2000). Laying nets over a crop and removing them for re-use can be labour-intensive, but a number of labour saving methods have been developed (Fuller-Perrine and Tobin, 1993; Taber and Martin, 1998; Duffy, 2000) and are now commonly used. Because of the fixed cost of the equipment required to apply and remove nets efficiently, it is more economical to use drape-over nets on large or high-value crops where bird damage levels are generally high (Fuller-Perrine and Tobin, 1993). On small or low-value crops the value gained from avoiding the damage may not outweigh the cost of netting.

Alternative drape-over netting options include one-, two- , four- or six-row netting or a 'lockout' system, whereby nets are draped over orchard trees or vines and then joined together to create a complete cover. The 'lockout' method requires more labour but less material, as the netting does not drape to the ground on the inside rows of the block. When spraying for botrytis and other diseases or to reduce fruit splitting, some growers use small tractors to enable them to spray underneath the netting, particularly for 'lock-out' and multiple-row netting systems. Less netting is also required when covering multiple rows, rather than a single row, and this improves cost-effectiveness.

Some growers construct lightweight total-exclusion netting systems, using second-hand water pipe for poles, star-droppers for anchors, and soft wire or baling twine to hold up low-cost, short-lived nets, such as fish gill nets. These systems have a high maintenance component and usually require replacing every one to three years. For crops that need only short-term protection, these lightweight systems may be appropriate if a low-cost source of labour is available for maintenance.

In Australia, permanent total exclusion systems are a popular form of bird exclusion for some tree crops. The basic design is simple, involving a pole and wire or cable structure supporting roof and side netting. Most structures now consist of panels of net that are individually erected, with each panel stretched tightly between wires joined at the selvedged edges. The perimeter poles are usually wood, although steel can be used. The structures are designed so that loads that develop on the structure from wind, rain, hail or snow are transferred back to the ground anchors guying back the perimeter poles.

According to netting manufacturers, some black nets have life expectancies of over ten years, and white nets last five to eight years. The supporting structure should outlast several nets with minimal maintenance if it is well designed and erected. In New Zealand, some wire netting has lasted even longer (45 years) and is resistant to chewing and breaching by birds.

Permanent netting may not be feasible for older established orchards or for crops planted on steeply sloping ground. Even where netting is technically feasible, it is a significant expense to purchase and erect. The most economical option is to incorporate the costs of design and erection of full netting into farm plans at the early establishment stage. There are considerable economies of scale as the area netted increases. An awareness of the main species responsible or potentially responsible for damage is necessary to determine the appropriate mesh size. Where bird damage is most severe around the edges of a vineyard, Taber and Martin (1998) suggest it may be worthwhile netting just the edges of a large crop.

Effects of netting on production and management

Netting can benefit fruit quality by reducing the prevalence of blemishes, sunburn and wind rub. Netting can, however, also increase the vigour of foliage and affect the size and colour of fruit by altering the microclimate. For example, hail netting, which has a much finer mesh (two millimetres) than that required for birds, reduces light levels by 20%–25% if black, 18% if grey and 12%–15% if white (Middleton and McWaters, 1996). Under hail netting, wind speed can also be reduced by up to 50% and humidity can increase by more than 50%. Despite a perceived change in temperature under nets, netting has little or no effect on temperature and does not offer frost protection (Middleton and McWaters, 1996).

Altered conditions under netting are likely to necessitate changes to management practices to ensure maximum productivity. Changes in the choice of rootstock and in pruning and irrigation practices may need to be considered, especially in the case of vigorously growing varieties. Disease management may also require further consideration in cooler climates, for slow-ripening varieties, and in disease-prone regions.

In Middleton and McWalter's (1996) study of the effects of hail netting in apple orchards in Stanthorpe (Queensland), Orange (New South Wales), and Drouin (Victoria), less fruit was produced under netting. Reductions in fruit set were not large and were beneficial in this study, as less thinning was required. Reduced fruit size and increased shoot growth occurred on vigorous trees under netting. The effects on fruit colour depend on the variety and fruit position. Pollination may also be affected by netting; fewer bees are observed on trees under black netting than on uncovered trees (Middleton and McWaters, 1996). Exclusion of insects has also been considered a benefit in orchards, for example by limiting fruit-fly damage to stone fruit (Lloyd et al., 2005). Placing beehives in the netted areas may overcome poor pollination.

For low-chill stone fruits, exclusion netting (hail net of two millimetres hole diameter) was found to enhance fruit development by seven to ten days and to improve fruit quality by increasing sugar concentration by 20%–30% and increasing colour intensity by 20% (Lloyd et al., 2005).

Other methods of exclusion

Attempts have been made to protect crops by using monofilament lines strung over crops. Knight (2000) found that birds were repelled about 25 centimetres from filament erected like a tepee over fruit trees. However, a field experiment to test monofilament lines placed at 30-centimetre intervals over a grape crop showed that they were ineffective in preventing damage by starlings and other species (Steinegger et al., 1991). There are also welfare concerns with monofilament lines, as injuries to birds can occur.

In a field experiment, Chambers (1993) demonstrated that covering individual table grape bunches with polyester sleeves significantly ($P < 0.05$) reduced the damage caused by Cape sparrows (*Passer melanurus*). The polyester sleeves did not reduce grape quality or yield. The obvious disadvantage of this approach is that it is labour-intensive and hence

costly. Although damage was reduced to almost negligible levels in this experiment, not all bunches were covered, so the birds had access to uncovered grapes. It is possible that if all bunches were covered the birds would have pecked through the sleeves, as occurred in one instance.

2.3.5 Chemical repellents

Chemical repellents (or deterrents) are aversive substances that are usually sprayed onto crops because their taste, smell, colour or physiological effect makes the treated fruit unattractive to birds (Mason and Clark, 1997). Many chemicals used or tested as bird repellents were originally registered as agricultural products such as insecticides or fungicides (Clark, 1998). Currently there are few available chemical repellents that can be use to prevent loss of fruit caused by birds. One limitation is that chemical repellents can leave residues in fruit that make them unsuitable for human consumption (Porter et al., 1996b). In addition, some chemical repellents are phytotoxic and damage sprayed plants. A further problem is the small size of the market for such chemicals. The cost of obtaining and keeping registration of agricultural chemicals has meant that neither industry nor government is prepared invest in minor-use chemicals such as bird repellents.

Primary repellents

Primary bird repellents are agents that produce an immediate avoidance response by birds because of their unpleasant smell or taste, or because they cause irritation or pain (Clark 1998). Considerable work has been conducted in the United States in the last 20 years on primary chemical repellents to protect agricultural crops from birds (Avery, 1992; Cummings et al., 1994; Curtis et al., 1994; Cummings et al., 1995; Avery et al., 1996b; Watkins, 1996; Watkins et al., 1996; Cummings et al., 1998a; Cummings et al., 1998b; Dolbeer et al., 1998; Gill et al., 1999; Askham, 2000). Much of this work has focused on methyl anthranilate, a human food-flavouring additive that occurs naturally in many plants. This work showed that some formulations of methyl anthranilate are effective in reducing bird damage to some horticultural crops, but that their effectiveness is variable.

In Australia, Sinclair and Campbell (1995) conducted cage trials testing the repellency of methyl anthranilate on four species of pest bird: the Adelaide rosella, silvereye, little corella and starling. They found that when alternative food was provided methyl anthranilate was highly repellent to all four species. However, field trials with the chemical

on apricots, grapes, cherries, and apples did not demonstrate effective repellency at application rates that were not phytotoxic (Sinclair and Campbell, unpublished). Staples et al. (1998) found that the chemical was phytotoxic to rice seeds and seedlings and also warned of its potential toxicity to marine animals if the chemical was applied in marine environments.

Naphthalene and capsaicin, although marketed in the United States as bird repellents, have not been shown to be effective in deterring birds (Dolbeer et al., 1988; Mason et al., 1991a; Clark, 1997). Mint derivatives (Avery et al., 1996a) and caffeine (Avery et al., 2005b) are other repellents that have undergone preliminary testing. However, field investigations have not been conducted.

Secondary repellents

Secondary repellents work by making birds feel ill, so that they subsequently develop a conditioned aversion to the food to which the repellents have been applied (Clark 1998). Methiocarb (Mesurol-75®) is a carbamate insecticide that is also used as a snail and slug poison. It acts by inhibiting the activity of acetylcholinesterase, an enzyme that catalyses the breakdown of the neurotransmitter acetylcholine. In the 1970s methiocarb was trialled in Australia as a bird repellent. It provided good protection against blackbird and silvereye damage over two seasons in trials in the Riverland region of South Australia, with the yield harvested from treated areas being almost double that of untreated areas (Bailey and Smith, 1979).

Porter (1982) compared the effectiveness of methiocarb and netting individual trees to protect sweet cherries from exotic bird species in New Zealand. The pest species present were mynas, starlings, blackbirds and song thrushes. Spraying with methiocarb significantly ($P < 0.001$) reduced damage: sprayed trees lost 10% of their fruit to birds, whereas unsprayed trees lost 80%. Sprayed and netted trees lost only 2% of their fruit to birds. Over a 12-year repayment period, spraying alone, without the high cost of netting trees, gave better financial returns. Methiocarb residues on the fruit were reduced by 50% (to within the New Zealand Agricultural Chemical Board limit of seven parts per million) after the fruit had been washed in water. Residues were reduced by 66% after washing in dilute detergent (Porter 1982).

Tobin et al. (1989a) tested the effectiveness of methiocarb spray in protecting cherries from pest birds (mainly starlings, American robins, and house and common grackles). Although they found that sprayed blocks had significantly ($P = 0.03$) less damage (6.5%) than unsprayed blocks (8.8%), the level of reduction in damage was not sufficient to justify the cost of spraying. In a later field trial, Tobin et al. (1991) found that spraying cherries with methiocarb did not significantly ($P > 0.5$) affect the average percentage of cherries damaged by starlings and 14 other species of birds.
In aviary trials, Cummings et al. (1998b) found that spraying with methiocarb significantly ($P < 0.01$) reduced the consumption of lettuce seedlings by horned larks (*Eremophila alpestris*). Topical application of methiocarb to sprouting tomato seedlings reduced skylark (*Alauda arvensis*) damage to minimal levels (Anonymous, 1970).

Hardy et al. (1993) conducted field trials to assess the safety of spray applications of methiocarb. They concluded that even heavy repeated spraying did not pose a hazard to wildlife, despite the fact that mammals and birds were exposed to the compound.

Methiocarb as a seed-dressing has had mixed results and is not generally recommended. Holding (1995) applied methiocarb to canola seed and recorded good deterrence against skylarks with a doubling of the yield in treated versus untreated plots. However, delayed germination may cause insufficient chemical to be absorbed by the sprouting seedlings, which may lead to increased bird damage. This has been found in the case of treated tomato seed (Bergman, 1970).

Porter and McLennan (1995) tested the effectiveness of cinnamamide (a secondary plant compound) and netting for protecting grapes from pest birds. The pest species present were mainly sparrows, silvereyes, greenfinches, blackbirds and song thrushes. Both treatments significantly ($P < 0.01$) reduced the numbers of pecked and missing grapes. Cinnamamide reduced damage by 40% and netting by 84%; however, neither treatment significantly increased mean bunch weight or mean yield. This was possibly because the vines compensated for missing grapes by increasing the size of the surviving fruit. Porter and McLennan (1995) found residues of cinnamamide in wine made from treated grapes, and this chemical also left a ‘plastic-like’ flavour on grapes, making them unacceptable for making wine.

Other secondary bird repellents that have been used in North America are lindane (an insecticide that stimulates the central nervous system) and captan and thiram (originally fungicides), which depress the central nervous system (Clark 1998). There is also Kocide®, which is a copper-based fungicide (Avery et al., 1994b); and fipronil, an insecticide developed for use on rice seed and other crops (Avery et al., 1998). Brugger et al., (1993) and Martinez del Rio et al., (1997) suggested that sucrose (household sugar) in high concentrations on fruit might act as an effective secondary repellent for starlings and other pest birds because they lack the enzymes necessary for its digestion. Avery et al., (1995a) found that caged starlings and cedar waxwings (*Bombycilla cedrorum*) consumed significantly ($P < 0.1$) more artificial fruit containing hexose (a mixture of glucose and fructose) than artificial fruit containing sucrose, and this preference overrode pre-existing preferences for fruit colour. However, tests conducted by Askham (1996) on starlings do not support the theory that birds are intolerant to sucrose.

Anthraquinone, commercially known as Flight Control®, is a polycyclic aromatic hydrocarbon that occurs naturally in insects, plants and fungi. Although commonly used in the manufacture of dyes and as a catalyst in the paper industry, this chemical has also been used as a grazing repellent to deter birds (particularly Canada geese) from golf courses, airports, urban and industrial areas and landfills, and as a seed coating and repellent to protect crops. Anthraquinone and related compounds have been shown to reduce consumption of rice, millet, sorghum and maize by red-winged blackbirds, brown-headed cow-birds and dickcissels (*Spiza americana*) (Wright 1962; Avery et al., 1997; Dolbeer et al., 1998; Avery et al., 2001; Cummings et al., 2002). Cage trials with horned larks indicated that high levels of damage (60%) still occurred to treated lettuce (York et al., 2000). However, York et al., (2000) suggested that bird damage was artificially high because of the nature of the enclosure situation and indicated that field trials were required.

Delivery of primary and secondary repellents

Repellents that are consumed target oral receptors if they are primary repellents, or gastrointestinal receptors if they are secondary repellents (Clark 1998). Chemical repellents are rarely delivered in raw form, but are combined with other substances and applied in accordance with label instructions (Clark 1998). Carriers, spreaders, stickers and wetting agents improve the deposition of the repellent. These products ensure even

coverage and improve retention by slowing environmental degradation and weathering losses. The stability of the repellent can be affected by carriers, stabilisers, solvents, binders, biocides and antioxidants (Clark 1998). The concentrations of the repellent agent and additives are important, as these will influence efficacy and cost. For some agents, concentrated applications can leave unacceptable residues. If toxic repellents are used, concentrated applications can cause blemishes on the crop, damage the foliage, or kill non-target species (Staples et al., 1998).

Tactile repellents

Clark (1998) investigated the use of contact tactile repellents applied to perches to irritate birds' feet. Starlings avoided perches painted with tactile repellents containing plant extracts or methiocarb. None of the substances tested caused illness in birds. Clark (1998) concluded that further work was needed to see whether such non-lethal repellents are useful for pest bird control.

A number of non-toxic, sticky or oily substances are used for bird control (Clark 1998). When applied to surfaces where birds perch, they avoid them. Some problems may occur with short-legged species (for example, welcome swallows, *Hirundo neoxena*, whose wing-tips sometimes become glued to the surfaces to which the gel has been applied.

Seed coating

Coating seeds with substances such as clay, cement, plaster (Dolbeer and Ickes 1994), diatomaceous earth (containing sharp particles), or starch can make it more difficult or unpleasant for birds to crack them open, thus reducing damage. Handling time increases, making the seeds less attractive to the birds. In cage tests, Cummings et al., (1998b) found that coating lettuce seeds with clay significantly ($P < 0.01$) reduced seed consumption by horned larks. These treatments have the potential to reduce damage to newly sown crops.

2.4 Benefits of birds

Birds can also provide many economic and environmental benefits including the control of insects, competition with, or predation of, pest birds and enhancement of environmental health and aesthetics. Many birds found in horticultural crops are insectivorous, including honeyeaters. These species may play important roles in controlling insect pests. For example, the most important factor influencing the mortality of the codling moth (*Cydia*

pomonella) is predation of the caterpillars by birds in autumn (Chapman et al. 1992). Birds are known to consume soil insects such as cockchafers and underground grass caterpillars (Subfamilies: Melolonthinae and Scarabaeidae), as well as codling moth pupae and the light brown apple moth (*Epiphyas postvittana*). Results show that bird predation can reduce grasshopper densities by 30%–50% (Joern 1986; Fowler et al. 1991; Bock et al. 1992). An integrated approach to managing birds and insects is likely to provide ongoing benefits in terms of reduced insect damage and reduced pesticide use. In some cases, insecticide spraying has been shown to increase the number of insect pests by inadvertently removing natural predatory insects (Prischmann et al. 2005). Birds also regulate harmful insects (Strong et al. 2000; Sanz 2001; Tremblay et al. 2001; Mols and Visser 2002).

In a study in Spain, caterpillar damage to oak leaves was significantly less at sites where breeding birds were encouraged, compared with control sites (Sanz 2001). In another study, bird predation reduced pest insects by 50% and resulted in a 30% increase in the growth of oak trees in the Missouri Ozark deciduous forest (Marquis and Whelan 1994). In Canada and Europe, birds have been shown to benefit orchards by controlling overwintering Lepidoptera (Solomon and Glen 1979; MacLellan 1971). In a study in northern Sweden (Atlegrim 1989) the total density of insect larvae was 63% lower where birds had access to larvae than where exclosures were used; this resulted in significantly less insect damage to the annual shoots of bilberry. In the Netherlands, great tits (*Parus major*) have been shown to reduce caterpillar damage to apple orchards (Mols and Visser 2002, 2007). In Mils and Visser's (2007) study in areas with breeding great tits, apples had 50% of the caterpillar damage of the control areas.

However, the ability of birds to regulate insect populations and reduce insect damage depends on a number of variables, including bird population density (East and Pottinger 1975), insect life cycle (East and Pottinger 1975), habitat (Belovsky et al. 1990) and insect population dynamics. Despite feeding on harmful insects, birds in some situations may have a negligible effect on insect populations or the damage they cause (East and Pottinger 1975; McLennan and MacMillan 1983).

Birds of prey and species that compete or exclude pest birds are desirable in horticultural settings. For example, magpies (*Gymnorhina tibicen*) are territorial and occasionally display agonistic behaviour towards, and attack, pest birds including sparrows (Barr 1986;

Morgan et al. 2006), starlings (Morgan et al. 2006) and sulphur-crested cockatoos (Cilento and Jones 1999). Raptors (Accipitriformes and Falconiformes), particularly sparrowhawks, goshawks, falcons and hobbies, are known predators of a range of pest birds. Attracting these birds to crops might provide economic benefits by reducing the numbers of pest birds and the damage they cause. However, providing habitat to attract desirable birds requires careful consideration and management.

Controlling bird populations may also have other unintended consequences to ecosystem function. For example disturbance and a decline in biodiversity, may increase risks of infectious diseases (Derne et al. 2001; Mills 2006). This has been attributed to the 'dilution affect' (Ostfeld and Keesing 2000). However, several other factors may be more important, and overide the effects of biodiversity on disease prevalence, e.g. the traits of particular species (Schmid and Ostfeld 2001).

2.5 Priorities for future research on pest birds

In contrast to the situation with pest mammals (e.g. Lever, 1985; Putman, 1989), there are fundamental deficiencies in our knowledge of pest bird species, their impacts, how to measure them and the costs and efficacy of commonly used management practices.

There are few techniques available for measuring impacts of birds (Dehaven, 1974a; Dolbeer, 1975; Dehaven and Hothem, 1979; Nemtzov, 2004) and these are time-consuming, or can be unreliable and inaccurate. Realistic economic assessments of damage and of social and environment costs are required before investments are made to manage perceived problems. Robust information on the extent, pattern and cost of bird impacts and the effectiveness of control methods is required. These data are essential for identifying the main species involved, identifying industries and regions most at risk, and assessing the benefits of bird control.

Measuring health-related impacts of birds usually involves estimates of economic and social impacts after epizootics or epidemics have occurred. However, preventative measures will require an understanding of the likelihood and risks posed by future disease outbreaks. There have been global increases in incidents of emerging diseases (Jones et al., 2008) that threaten the economy, environment and human and animal health. Wildlife is source for the majority of these (Jones et al., 2008). An understanding of epidemiology of

viruses in wildlife and their interaction with humans and livestock are important in assessing their risks. There has been a significant increase in surveillance for diseases in wildlife, particularly in wild birds. However, broad-scale surveillance is logistically difficult and costly because of natural low prevalence of viruses, and wide variety and abundance of potential hosts. Improvements are needed in targeting surveillance according to associated risks, and to improve efficiency.

Current techniques used for managing birds have rarely been rigorously evaluated in terms of their ability to reduce abundance or impact. There is high variability in bird crop systems between bird species, their abundance (Dyer, 1967; Tracey et al., 2001; Tracey et al., 2007), and the extent of damage they cause (Wiens and Dyer, 1977; Whitehead et al., 1995). Large samples sizes are therefore required to confident assess management treatments. For example, in studies of birds in vineyards, >80 replicates were required to ensure that a 10% reduction in damage is detected (Tracey et al., 2001). There are no known evaluations of bird management methods with this number of replicates. Empirical studies with sufficient sample sizes are required to assess even the most commonly used techniques for managing birds, including shooting, trapping, netting and acoustic and visual deterrents.

PART A: ECONOMIC IMPACTS

Preface

Part A considers the economic impacts of birds using bird damage to fruit as a case study. In Chapter 3 I describe a method for measuring bird damage in wine grapes. In Chapter 4 I evaluate the efficacy of lethal (nest removal, shooting, trapping, poisoning) and non-lethal (scaring, netting) methods in reducing damage to fruit, using experiments to reduce bird damage and pest bird abundance.

3. A technique to estimate bird damage in wine grapes

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Abstract

Birds cause damage in many agricultural systems around the world. Measurement of such damage is an important first step in its effective management. We develop a visual assessment technique and a progressive sampling strategy using 5 strata and suggest sample sizes necessary to achieve an estimate of bird damage within a standard error of 5%. This strategy improved sampling efficiency by 67%, 79% and 80% compared to stratified systematic, standard systematic and random sampling. With an average cost of under $(AUS) 6 per block, this technique is a rapid inexpensive method to estimate bird damage to vineyards and has application to most crop-bird situations.

3.1 Introduction

Birds cause damage in many agricultural systems around the world (United States: De Grazio, 1978; Africa: Bruggers and Elliot, 1989; Europe: Mooij, 2001; Canada: Somers and Morris, 2002; Asia: Nemtzov, 2004; Australia: Tracey et al., 2007). Accurate and efficient damage assessment techniques underpin any research and management efforts to

reduce damage. Assessment techniques currently available to researchers and managers are either unverified, or are time consuming and therefore costly. Previous studies use standard random or systematic sampling procedures by counting individual fruits or plants (Nemtzov, 2004), or by weighing or visually assessing them (De Grazio et al., 1969; Stevenson and Virgo, 1971; Dolbeer, 1975; DeHaven and Hothem, 1979). In this paper we describe a visual assessment technique and progressive sampling strategy to estimate bird damage in wine grapes and discuss applications to other crops.

3.2 Methods

Random bunch selection

To avoid over-sampling of more visible bunches of grapes a technique for selecting bunches on each vine at random was used. A pole marked at 10 cm intervals was placed vertically in one of seven (0–6) locations along each selected vine. Random numbers were generated between 7 and 12 for the vertical axis and 0 and 6 for the horizontal axis. The vertical numbers corresponded to all harvestable bunches occurring between 70 and 120 cm above ground level. Grapes were grown within this height for all vineyards sampled, except one with lower trellises where a height of between 50 and 100 cm was selected. A horizontal number of 3 required placement of the pole at the vine stem; 0 at the left hand edge; 6 at the right hand edge; and 1 through to 5 at equidistance between the extremes (see Figure 3.1). The closest bunch to the pole was selected. Once the vine was located, one observer could locate and assess a bunch in approximately 10 seconds.

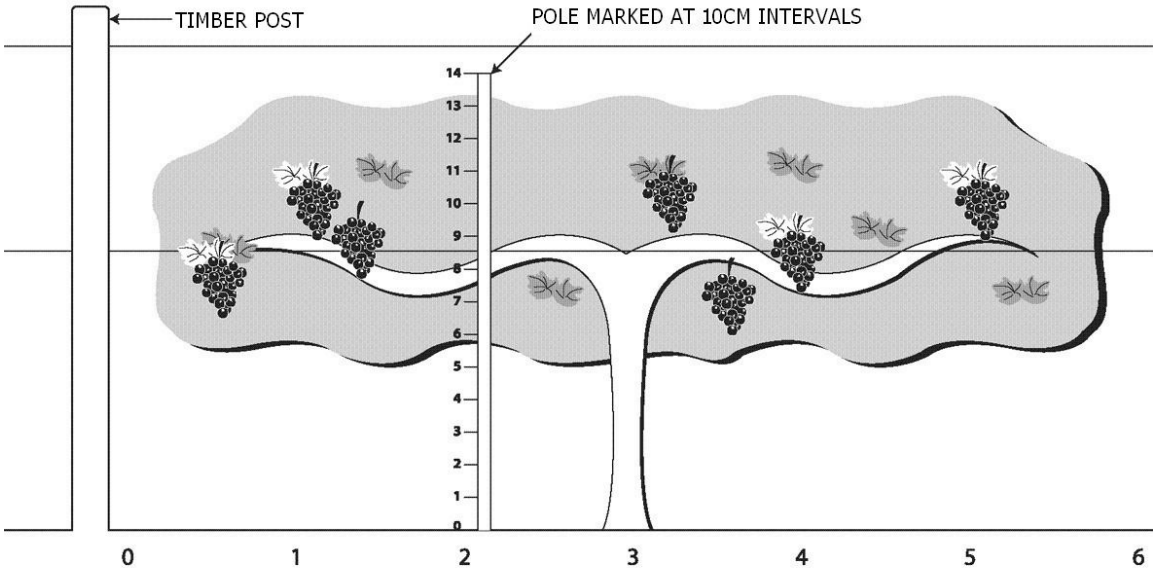


Figure 3.1. Technique for selecting random bunches of grapes for assessment of damage.

Visual assessment

Grape bunches (n = 26,500) were visually assessed by eleven observers to determine mean percentage bird damage. Visual estimates of bird damage to each bunch included both pecked and missing grapes, and were initially made to the nearest 1 percent, then to the nearest 5 percent if the damage estimate was between 10 to 90 percent, and to the nearest 1 percent otherwise, as assessment accuracy is higher at the extreme levels of damage. This overcame difficulties associated with ranking scales (DeHaven, 1974a). In an attempt to minimise error, observers practiced on bird-damaged bunches and used a chart of bunches displaying different levels of damage (Appendix I). Presence of disease was also recorded, and where possible damage was differentiated for different species. For example silvereyes (*Zosterops lateralis*), yellow-faced honeyeaters (*Lichenostomus chrysops*) and other small honeyeaters caused small punctures in the fruit; noisy friarbirds (*Philemon corniculatus*), noisy miners (*Manorina melanocephala*) and red wattlebirds (*Anthochaera carunculata*) caused larger angular punctures, with both these groups of species often hollowing out fruit. Crimson (*Platycercus elegans elegans*) and eastern (*Platycercus eximinius*) rosellas left small triangular marks made by their lower bill, and sulphur-crested cockatoos (*Cacatua galleria*) and galahs removed large chunks of fruit relative to their bill size, or for stone or pome fruits would split the fruit to access the seed (Section 4.2). Lorikeets (musk,

Glossopsitta concinna, little, *Glossopsitta pusilla*) were rarely observed in this study, but for stone and pome fruit left horseshoe-shaped marks made by the lower part of their bill and triangular marks made by the upper part of their bill (see Temby 2002 for a further description). Damage can also be caused by animals other than birds, including insects, flying foxes, mice, foxes, and dogs. Non-bird damage was excluded whenever identified. Insect damage from bees, European wasps, ants etc. occurred occasionally (41 of 32,381 bunches assessed), and was easily distinguished from bird damage. Flying fox (grey-headed, *Pteropus poliocephalus*, and little red, *Pteropus scapulatus*) damage was not observed during this study but is distinguished from bird damage by the teeth marks in the fruit and the size of the spats (fragments of skin and fruit pulp compressed together), which are much larger than the remnants of fruit dropped by birds (Temby 2002). Mouse (*Mus musculus*) damage was rarely observed (<0.1%, 13 of 32,381 bunches assessed), and identified by their teeth marks in the fruit. Damage to bunches close to the ground (30-50 cm, Figure 3.1) was also rarely attributed to foxes (*Vulpes vulpes*) and dogs (*Canis lupis familiaris*) (3 of 32,381 bunches assessed), where grapes were stripped from bunches from underneath and tracks and faeces were observed.

The visual assessment procedure was tested in the field by comparing visual estimates (n = 594, 8 observers) with actual percent damage. Actual percentage damage to individual bunches was calculated by counting the number of missing, pecked and remaining grapes on each bunch.

Initial sampling strategy

One hundred and twenty-nine blocks of grapes on nine properties were sampled for bird damage, with a block being a continuous planting of a single variety sampled in a particular time period. Thirteen grape varieties were sampled: six red and seven white. Sixty blocks were sampled immediately prior to harvest, twenty-one of which were also sampled between veraison, when grapes first change colour, and one week before harvest. The first and last rows from each block were sampled sequentially from a randomly chosen vine. Interior rows and vines were also systematically sampled. One bunch was selected from each interior vine and two bunches from all edge vines on sampled rows.

A new progressive sampling strategy was developed for estimating bird damage and compared with three other methods. The efficiency of the four sampling strategies were

then compared; (1) the progressive sampling strategy, (2) a stratified systematic sample using the same 5 proposed strata, (3) a standard systematic sample and (4) a random sample, necessary to achieve an estimate of damage, within a 5% standard error, to 261 vineyard blocks sampled. Cost of labour was assumed to be $AUS 18.26 per hour (Farm and/or Orchard Hand – Level 4 Casual: Tasmanian Industrial Commission, 2006).

3.3 Results

Evaluation of visual assessment methods

The majority of bird damage was by missing grapes (68.6 ± 1.5 %, n=11,384 damaged bunches), with 18.8 ± 0.80% of bunches pecked, and 12.5 ± 0.65 % of bunches both pecked and missing.

Despite training, observers under-estimated bird damage to individual bunches, particularly at mid percentages (40–60%). To allow correction of damage data, observer effects were treated as random and data was pooled for all observers. An inverse estimator for the calibration data was also used for simplicity in calculating confidence intervals (Armitage and Colton, 1998).

To determine a correction model, percentages of actual (X) and estimated (Y) damage were first logit transformed to linearise the response and to remove variance heterogeneity. By definition, $\text{logit}(Y) = \log(Y/(100 - Y))$.

The prediction model for $\text{logit}(X)$ is then $\text{logit}(X) = 0.708 + 0.811 \times \text{logit}(Y)$, or equivalently

$$X=100/(1+\exp(-[0.708+0.811\times\text{logit}(Y)]))$$

Development of a progressive sampling strategy

In all cases, damage inside a block was less than the damage observed on the boundary, except when overall damage was less than 5% (n = 129 blocks; Figure 3.3).

To estimate the mean percent damage for a block we assumed an equal number of bunches per vine in each block. A weighted average of the estimated means within each stratum was then determined, with the weights proportional to the number of vines in each stratum; $\sum_i d_i p_i$, where, for $i = 1,\ldots,5$, d_i = mean damage for stratum i and p_i = proportion of total number of vines in block that are in stratum i. To determine appropriate sample sizes we examined the standard deviation of the results versus the mean within each stratum (Figure 3.4). The least squares fit for the line (Figure 3.4) as: SD = α [Mean(100 − Mean)]β, where $\alpha = 0.079$ and $\beta = 0.778$.

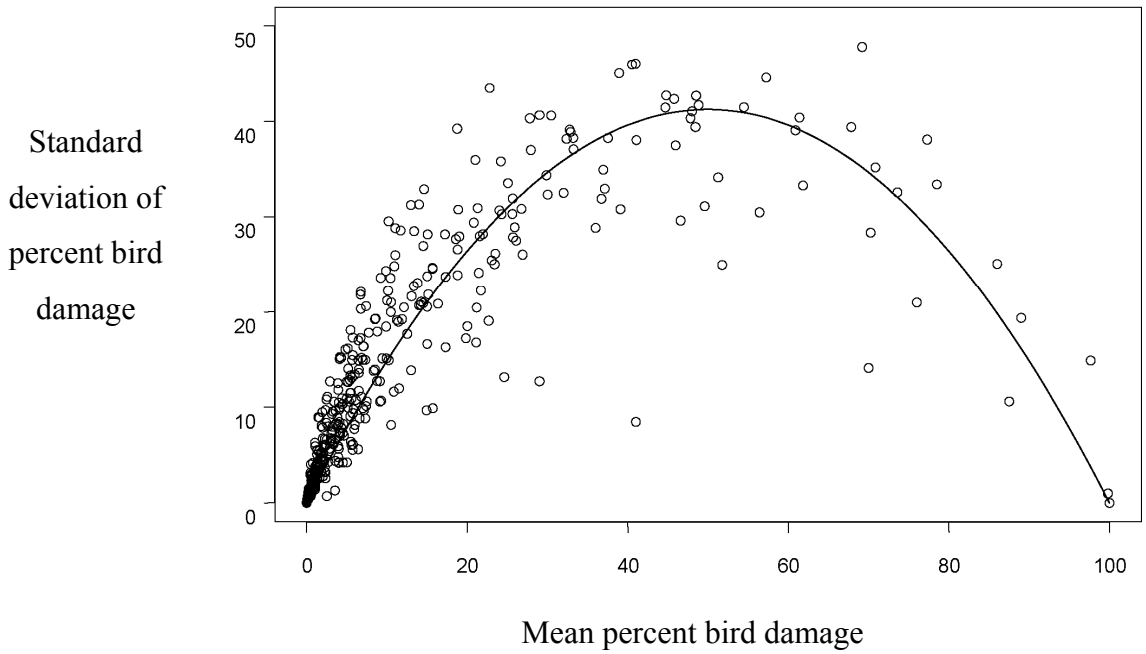


Figure 3.2. Standard deviation versus mean percent bird damage for each stratum (1-5).

Based on the above model for the variation of results within strata we can determine the minimum sample size needed to estimate the mean percentage damage within a stratum so as to place an upper bound on its standard error. For example, should the mean percent damage within a stratum need to be estimated with a standard error of 3% or less, then the sample size needed, n say, must satisfy:

$0.079\ [\text{Mean}\ (100 - \text{Mean})]0.778/\sqrt{n} \leq 2.$

If Mean is 20% say, then n must be at least 80. Figure 3.5 plots the minimum sample size versus mean for when the standard error of the mean equal 3%, 5%, 7% and 10%.

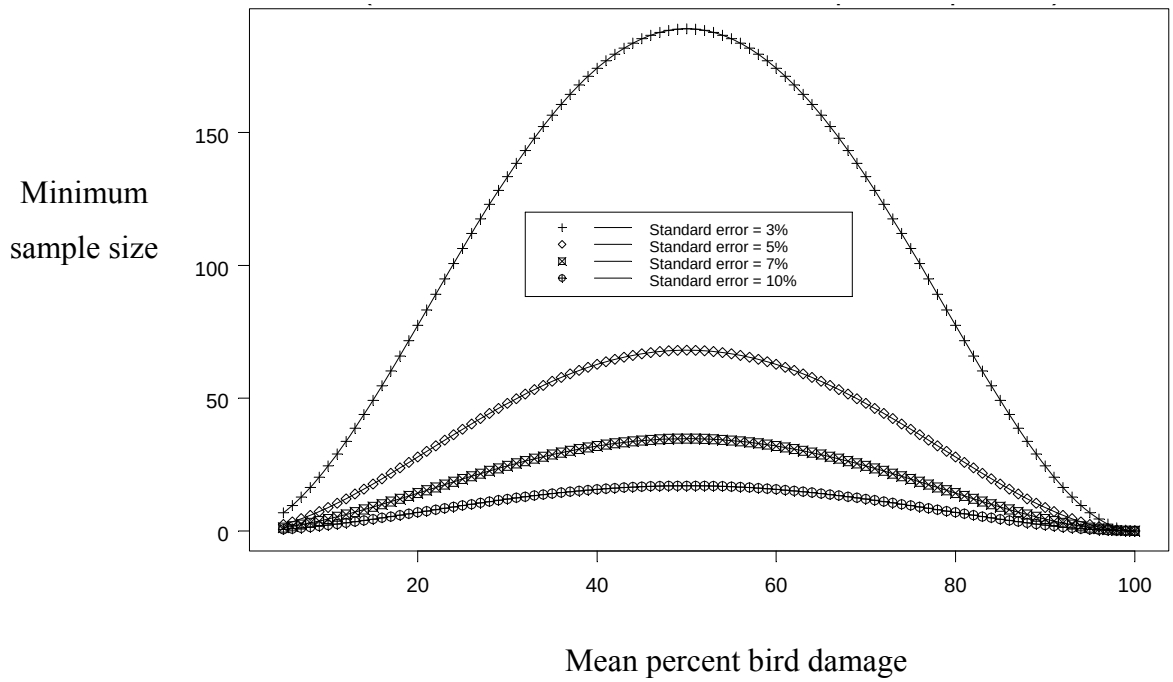


Figure 3.3. Sample sizes needed for estimating damage per stratum with standard error (3%, 5%, 7% and 10%).

When estimating the percentage damage of a block based on a weighted average of the mean damage within each of the separate strata, the standard error of the overall mean estimate will depend on the relative sizes of the strata. Let p_i denote the proportion of vines in stratum i (i = 1, 2,…, 5) relative to the total number of vines in all five strata and τ_i equal the corresponding standard error of the percent damage estimate in that stratum. Then the standard error of the estimated mean percent damage for the block, τ say, is given by:
$\tau = \sqrt{(\sum_i p_i 2 \tau_i 2)}$. Hence τ is influenced by the maximum p_i (i = 1, 2, …, 5).

The progressive sampling procedure is based on the results of Figure 3.5 for any desired standard error. We aimed at achieving a standard error of 5% and assumed the underlying percent damage was 10%. Hence n=10 vines was chosen from each outside stratum (1–4). If damage was less than 5% in any outside stratum, then no more sampling was necessary as we could be confident that overall damage was less than 5% (Figure 3.3). If any stratum was greater than 5% in any outside stratum, the interior of the block was also sampled in the same way (Stratum 5). If damage was greater than the assumed 10% in any stratum

then more samples were taken from that stratum relative to the estimated percent damage (Table 3.1).

Progressive sampling strategy: a new method

This section summarises the progressive sampling strategy, based on the information above. To estimate the mean percent damage within a vineyard each block was stratified into 5 strata (Figure 3.4).

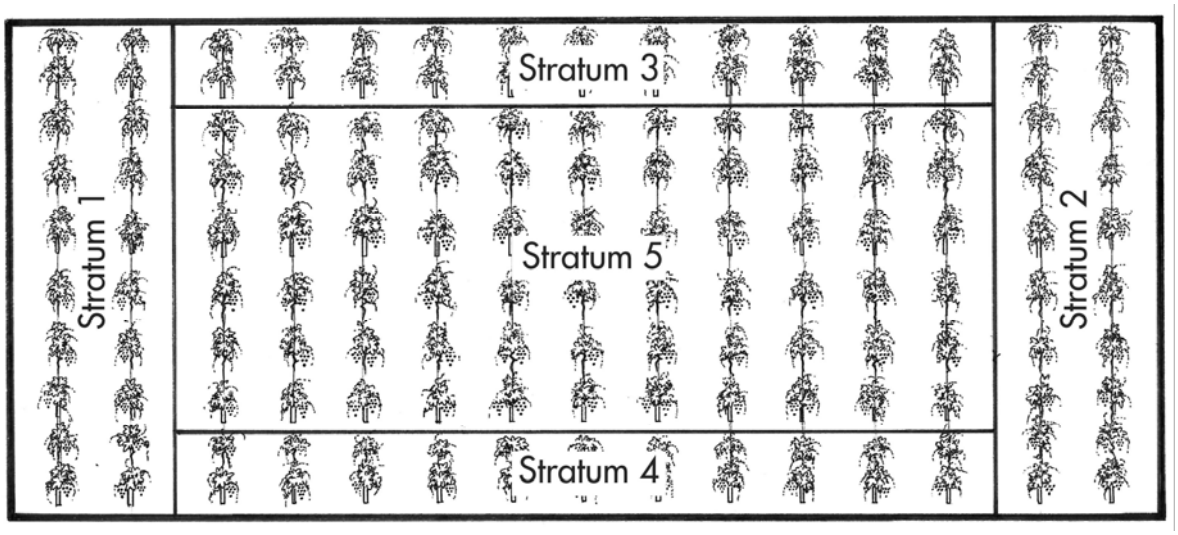


Figure 3.4. Stratification scheme for vineyard blocks adopted in this study.

The stratification scheme above is based on results showing that damage is more severe at the boundaries of the block (Figure 3.3), but not always uniform between boundary strata. For example, end rows of a block contained within rows of other grape varieties were not as severely damaged as outside rows adjacent to perching habitat. Hence the separate strata for each of the four boundaries.

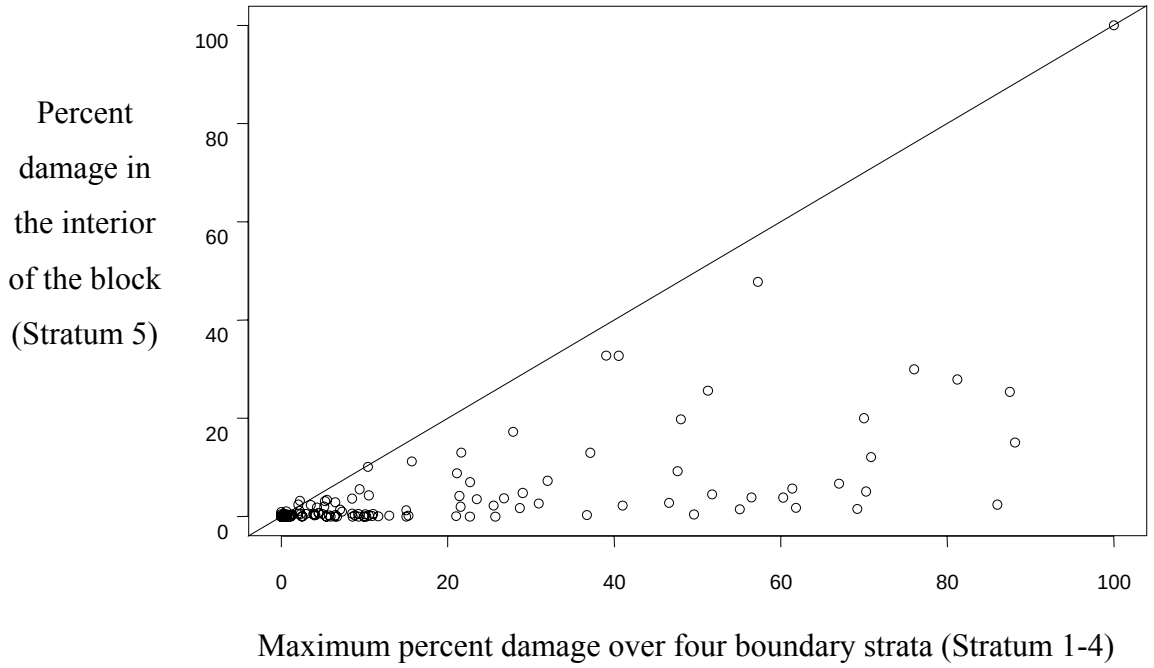


Figure 3.5. Bird damage (%) in the interior strata versus bird damage (%) to boundary strata within individual blocks of grapes.

For each block, mean bunch damage for bunches within each stratum was estimated separately. Here the percent damage per bunch was assumed to be a linear combination of overall mean percentage damage, a random component due to the vine and the bunch. These means, and associated standard errors, were estimated using ASREML (Gilmour et al., 2002).

One bunch was randomly selected from 10 systematically selected vines in each outside stratum (1–4) of sampled blocks. If mean damage exceeded 5% in any outside stratum 10 samples were taken from the interior of the block (Stratum 5). If damage was greater than 10% in any of the 5 strata, additional bunches were also sampled from those strata. In each case, Table 3.1 was used to determine the extra number of samples required.

Table 3.1. Sample sizes needed to estimate percent damage with 5% standard error (derived from Figure 3.5).

Bird damage (%)	5	10	20	30	40	50	60	70	80	90	95
Sample size	4	10	24	37	46	49	46	37	24	10	4

Efficiency of damage assessment techniques

The mean time taken to sample a block using the progressive sampling strategy was significantly less than when using stratified systematic, standard systematic and random sampling ($P < 0.001$, $df = 520$, t statistic = −46.4, −39.3, −20.2, Figure 3.6), improving sampling efficiency by 67%, 79% and 80% respectively. On average the cost of sampling a vineyard block using the progressive sampling technique was $AUS 5.77 ± $0.35 ($n = 261$, range $0.97–$18.95).

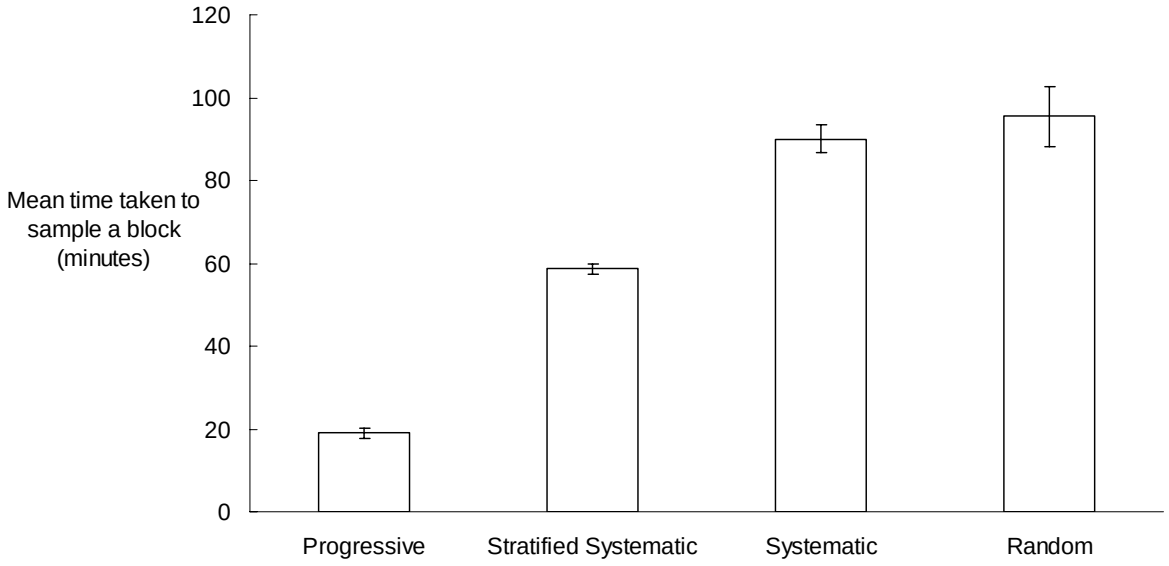


Figure 3.6. Mean time (minutes) taken to sample a block using progressive, stratified systematic, systematic and random sampling with 95% confidence intervals.

3.4 Discussion

Random bunch selection

Random bunch selection is necessary to avoid over-sampling of more visible bunches, which has been achieved previously using a combination of ropes, several poles or hoops and two or more observers (DeHaven and Hothem, 1979; Martin and Crabb, 1979). The selection procedure developed was simple and efficient in selecting random bunches. The same pole could be used in vineyards of any trellis height, provided random numbers were generated separately for different heights of vines. Once the vine was located, one observer could locate and assess a bunch in approximately 10 seconds. This was six to 18 times

more efficient than previous techniques which took between 30 and 60 seconds with two or three observers (DeHaven and Hothem, 1979; Martin and Crabb, 1979).

Evaluation of visual assessment methods

Despite training, all eight observers underestimated percent damage to selected bunches, particularly at mid percentages (40–60%). This emphasises the importance of calibrating visual estimates. Most other studies which visually estimated bird damage to wine grapes used either a damage class or a pre-transformed ranking scale (Table 3.2). In studies that compared visually estimated damage with known damage, most concluded that damage was accurately classified after a period of training. However, large classes were used (e.g. rank 1 = 0–5%; 2 = 5–20%; 3 = 20–50%; 4 = 50–80%; 5 = 80–95%; 6 = 95–100%, Stevenson and Virgo, 1971; Somers and Morris, 2002), and with the exception of Somers and Morris (2002), accuracy within classes was not reported.

We recommend estimating percent damage to individual bunches as described rather than using a ranking scale, as this is equally efficient, overcomes difficulties with uneven distribution of damage within classes (DeHaven, 1974b), and allows corrections of likely errors. If damage classes are to be used, we suggest testing the accuracy of classes and distribution of estimates within classes and using, where possible, bunches with actual rather than simulated damage for validation. This maybe particularly important where birds peck grapes as well as remove them. Our study suggests that pecked damage can be common (31.6 ± 1.03 % of bunches with pecked or both missing and pecked grapes; cf. <1%, Somers and Morris 2002). Where this occurs, we recommend training and validation with bunches containing both missing and pecked grapes.

Table 3.2 Type of assessments used to estimate bird damage to wine grapes.

Type of assessment	**Accuracy measured**	**Source**
Counting	NA	(Askham, 1992)
Counting	NA	(Toor and Ramzan, 1974)
Weighing	NA	(Porter and McLennan, 1995)
Ranking scale, counting and weighing	No, NA	(Hothem and DeHaven, 1982)
Percent estimate	No	(Chambers, 1993)
Percent estimate	No	(Curtis et al., 1994)
Percent estimate	Yes (n = 594, 8 observers)	**This Study**
Ranking scale	Yes	(Martin and Crabb, 1979)
Ranking scale	No	(DeHaven 1974a)
Ranking scale	No	(Bailey and Smith, 1979)
Ranking scale	Previously tested	(Martin and Jarvis, 1980)
Ranking scale	Yes (n = 10, 85% of bunches scored within the damage class)	(Stevenson and Virgo, 1971)
Ranking scale	No	(Yim and Kang, 1982)
Ranking scale	Yes (n = 400, 2 observers)	(DeHaven and Hothem, 1979)
Ranking scale	Previously tested	(DeHaven and Hothem, 1981)
Ranking scale	Previously tested	(Hothem et al., 1981)
Ranking scale	Yes (n=104)	(Somers and Morris, 2002)

Progressive sampling strategy

This study found bird damage was always higher in at least one outside edge than in the interior of the block, except when overall damage is low (<5%). Higher damage on the edges of the crop is consistently observed for many bird species and crop situations. For example, starlings, cedar waxwings, *Bombycilla cedrorum*, and American robins, *Turdus migratorius*, in wine grapes (Somers and Morris, 2002); sulphur-crested cockatoos, *Cacatua galerita*, and galahs, *Eolophus roseicapilla*, in sunflowers (Fleming et al., 2002b); and grackles, *Quiscalus mexicanus*, in grapefruit (Johnson et al., 1989). The progressive sampling strategy is significantly more efficient than other methods of sampling ($P <$

0.001), with this difference likely to be even greater in larger blocks, and where there is a highly skewed spatial distribution of damage. This is commonly observed in bird-crop conflicts (corn: Dyer, 1967; wine grapes: DeHaven, 1974b; apples: Halse, 1986; cherries: Sinclair and Bird, 1986; rice and sunflowers: Subramanya, 1994). The progressive sampling strategy presented here is likely to provide similar or increased improvements in efficiency in most bird-crop situations. With an average cost of sampling a block under $AUS6, this strategy could be routinely implemented to improve bird damage management decisions in viticulture and other agricultural crops.

4. Assessing lethal and non-lethal methods for reducing bird damage to fruit

Abstract

Context:

(1) Birds are well known pests of fruit and other horticultural crops, and many lethal and non-lethal methods are used to reduce their damage. However, there is considerable variability in bird-crop systems and measurements of damage and abundance are rarely used to evaluate the efficacy of control methods.

Aims:

(2) We estimate pest bird abundance in orchards and vineyards and evaluate the efficacy of lethal (nest removal, shooting, trapping, poisoning) and non-lethal (scaring, netting) methods in reducing damage to fruit.

Methods:

(3) A field study was conducted across 101 orchards and vineyards over seven years to test the efficacy of netting, shooting and scaring treatments in reducing bird damage. Generalised linear mixed model analyses were used to test for the effects of these treatments when used either individually or in combination, on percent bird damage across 185 property year records.

(3) The short-term effects of nest removal (using systematic searches and breeding data in 9 vineyards in one breeding season and 1 vineyard over four breeding seasons) and shooting on the abundance of starlings; and the effects of trapping (76 trapping events, 21 seasons, 22 sites) on the abundance of starlings, blackbirds, silvereyes and sparrows was also investigated in fruit crops. Bird abundance was estimated using variable circular plots and a Horvitz-Thompson type estimator to account for differences in detection with distance, species, group size and activity.

(4) Consumption of novel food types and estimated proportions of birds feeding were used to evaluate the feasibility of poisoning for starlings and corvids. The main factors influencing the proportion of birds captured and the number of birds feeding were investigated using generalised linear mixed models.

Key results:

(5) One-hundred and forty bird species were observed in vineyards and orchards of which 29 native and 7 introduced species damaged fruit. Starlings were the most common pest (70.4%, mean density 419± 112 starlings/ km^2). Netting was the most effective treatment in reducing damage, but birds breached nets through holes or gaps and consumed fruit through netting, with damage as high as 56± 4.8% recorded under netting. Shooting (20.5± 3.8%) was not as effective as netting (10.7± 2.8%), but less than a third of the cost ($538 vs $1,903/ha/property) and had 13% lower damage compared to nil treatments (33.2± 5.6%). This was likely to be a result of scaring birds away from the crop, as the number of birds shot was unrelated to damage caused and the numbers shot were low in relation to population size (35.0±7.9%). However, scaring with electronic devices and visual deterrents had no effect on bird damage, indicating that although birds may respond initially to scarers, they quickly acclimatise to new stimuli that do no pose a physical threat. Property size was significant in the final model with smaller crops more susceptible to damage, but crop type (grapes, cherries, apples, pears) and control effort had no affect on damage.

(6) Systematic nest searches indicate that nest removal would be highly effective (82%) in limiting starling reproduction, but other methods of lethal control had limited success (poisoning, shooting, trapping), had likely but un-measured non-target risks (poisoning, trapping) and were expensive ($25-$133 / starling). Overall effectiveness of trapping was low (5%) but varied with species and trap type.

Conclusions and Implications:

(7) Despite their widespread use, lethal methods used to control starlings, blackbirds, sparrows, silvereyes, and corvids had limited effectiveness for reducing pest populations in vineyards and orchards. However, shooting as a method to scare birds from the crop, rather than to control populations, was effective in reducing damage. In addition, results suggest nest removal would significantly reduce reproduction and could have benefits for the long-term management of starling populations.

(8) Netting was the most effective method in reducing damage and has considerable advantages when large numbers of native species are present. However, damage under netting can be significant and should be considered before investing in nets. Selecting

appropriate netting for the species involved, avoiding nets in direct contact with fruit and improved applications and maintenance of netting is likely to improve their effectiveness.

(9) The effectiveness of non-lethal and lethal controls was highly variable between control methods and combinations, bird species, abundance, trap types, and food presented. Large sample sizes and/or consideration of the variability in effectiveness, bird species, abundance and damage over time and between sites are recommended in evaluations of bird management methods.

4.1 Introduction

Around the world bird pests such as starlings (*Sturnus vulgaris*) (Conover and Dolbeer, 2007), mynas (*Acridotheres tristis*) (Dawson and Bull, 1970; Dhileepan, 1989), blackbirds (*Turdus merula*) (Moran et al., 2004), wood –pigeons (*Columba palumbus*) (Murton and Jones, 1973), corvids (Corvidae) (Weatherhead et al., 1980; Sinclair, 1998), silvereyes (*Zosterops lateralis*) (Rooke, 1983), honeyeaters (Meliphagidae) (Tracey et al., 2001), lorikeets (Loriinae) (Tracey et al., 2007), rosellas (*Platycercus* spp.) (Fisher, 1991), cockatoos and corellas (Cacatuidae) (Environment and Natural Resources Committee, 1995) are known to cause significant damage to horticultural crops. Some of these species also cause damage to cereal crops (Dolbeer et al., 1978), feedlots and grain storage areas (Feare, 1975); and are potential hosts of parasites and diseases (Weber, 1979). Pest birds also have environmental impacts, for example, they can prey upon seabirds and their eggs (Byrd, 1979; Watson et al., 1992) and usurp native hole-nesting birds (Green, 1983; Pell and Tidemann, 1997). However many are also native and locally protected species and can have ecological and agricultural benefits (Tracey et al., 2007; Triplett et al., 2012).

The most commonly used methods to reduce impacts of birds include netting (Draulans, 1987), scaring (Bomford and O'Brien, 1990), shooting (Murton et al., 1974; Dolbeer et al., 1993) and trapping (Weatherhead et al., 1980; Conover and Dolbeer, 2007). Chemicals are also used occasionally to repel or poison birds (Spurr, 2002). Netting is increasingly used to protect orchards, and despite high up-front and on-going costs (Slack and Reilly, 1994), it can be cost-effective in vineyards where bird damage exceeds 15% (Tracey and Vere, 2007). While scaring with visual or acoustic devices, is usually ineffective (Bomford and O'Brien, 1990; Bishop et al., 2003; Tracey et al., 2007 for review).

Lethal controls for pest birds are often used, but most attempts fail to reduce pest bird populations or damage. For example, shooting to reduce wood pigeon impacts in Cambridgeshire UK (Murton et al., 1974), organophosphate for controlling quelea (*Quelea quelea*) in Africa (Ward, 1979), PA-14 (a-Alkyl [Cll-C15]- omegahydroxypoly [oxyethylene], a non-ionic avian lethal surfactant with wetting characteristics) to control starlings and red-winged blackbirds (*Agelaius phoeniceus*) in Tennessee, North America (White et al., 1985), and explosives to reduce starling damage to cherries (Tahon, 1980) all failed to achieve management objectives. Feare (1991) suggests two main reasons for this, (1) that pest birds usually have a wide geographic range and much of the population is inaccessible to control and (2) that control attempts can be counter-acted by compensatory increases in breeding and survival.

In comparison, the use of lethal control where most of the population is accessible, where the species has low reproductive rates, (e.g. laughing gulls, *Larus atricilla*, Dolbeer, 1998), or where they are highly concentrated can result in large population reductions in the short-term (Besser et al., 1967; West, 1968; Besser et al., 1984; Blanton et al., 1992) and long-term (Porter et al., 2008a).

The effectiveness of poisoning and most trapping programs depends on the attractiveness and acceptability of novel food types to pest birds and other non-target species. Feeding behaviour and diet of pest birds have been well documented (Marchant and Higgins, 1990; Higgins, 1999; Higgins et al., 2006) and there have been several studies investigating the cues for detecting and assessing food. For example, while birds use colour to select food, they can also adapt their visual choices over time according to taste (Cowe and Skelhorn 2005). They also develop preferences for, or aversion to foods using social cues (Mason and Reidinger 1981, 1982; Mason et al. 1984), with flocking species more successful at locating quality food, because individuals use feeding by others as a cue to detect and select food (Ekman and Hake 1988).

In North America and New Zealand avicides are currently applied to cereal grains, cereal pellets, bread and dripping, sultanas and nuts for controlling starlings, red-winged blackbirds, corvids, ravens and gulls, and magpies (*Gymnorhina tibicen*) (Bull, 1965; Besser et al., 1967). Avicides are also used occasionally in Australia (alphachloralose, 1,4-aminopyridine and fenthion) for restricted purposes (Tracey et al., 2007). Effectiveness

(Porter et al., 2008a), community attitudes (Tracey et al., 2007), impacts on non-target species (Bruggers et al., 1989; Cummings et al., 2002; Custer et al., 2003), animal welfare concerns (Ministry of Agriculture Fisheries and Food, 1997); and residual or secondary effects in the food chain (Cunningham et al., 1979; Schafer, 1984) require careful consideration before avicides are considered. Preferences of non-target species to different food types are an important consideration as most avicides are toxic to all bird species. DRC-1339 (3-chloro-4-methylaniline hydro chloride, Flockoff® or Starlicide®) is an exception, where starlings, corvids and gulls are highly sensitive (Eisemann et al., 2003); while mammals (except cats) and many native North American birds are not susceptible (Eisemann et al., 2003).

This paper will evaluate the efficacy of lethal (shooting, trapping, poisoning, nest removal) and non-lethal (scaring, netting) methods in causing short-term reductions in pest abundance and damage to fruit.

4.2 Methods

Study site

The study was conducted in vineyards and apple, pear and cherry orchards of the Orange region of New South Wales (33.3° S,149.0° E). Sites are interspersed with eucalypts (*Eucalyptus macrorhyncha, Eucalyptus seeana, Eucalyptus tereticornis, Eucalyptus viminalas*), pine (*Pinus radiata*) plantations, mixed farming, and sheep and cattle grazing country. The area has a cool climate (mean temperature: 7° C to 18° C) with medium to high rainfall (mean annual rainfall: 920 mm). Vineyards and orchards range in size from 0.3 to 480 hectares, but the majority are less than 20 ha. Most vineyards have five or more varieties of grapes. The main types include cabernet sauvignon, cabernet franc, merlot, shiraz, pinot noir, sauvignon blanc and chardonnay. Orchards include cherries (Lapins, Sweetheart, Sunburst, Chelan, Merchant, Rons, Kordia, Simone, Regina), apples (Jonagold, Sun-Downer, Pink Lady, Braeburn, Granny Smith, Fuji) and pears (Beurré Bosc, Corella, Packham's Triumph, Williams, Sensation and Winter Nelis).

Bird abundance

Bird abundance was estimated using variable circular plots (Bibby et al., 2000). Fifteen experienced bird observers counted all species seen or heard for 10 minutes from fixed

locations at each site. Locations were selected using stratified random sampling, with vegetation type (vineyard or orchard, grassland, woodland, introduced vegetation) used to define the strata. Between 6 and 34 locations were selected on each site (not closer than 100 metres apart), with the same locations used for consecutive counts to maximise precision when comparing change of abundance over time. On average 54.7 (*s.d.* = 57.1) locations were sampled per season per site. Counts were conducted within the first 3 hours after sunrise, with the exception of those carried out during bait preference trials, which were completed each daylight hour. Numbers of individuals, distance, vegetation type, activity, wind direction and speed, and temperature were recorded to allow examination of their effects on the probability of detection (g) (n=153,352 birds counted, 27,044 individual sightings, 2,466 point locations, 344 count days, 101 sites). Predictions for the empirical detection, Y, were obtained from Poisson generalised linear models for significant variables (P<0.05), which included distance, species, group size, activity and first order interactions. The final detection models were obtained by including distance, spline of distance, species and activity in two models, determined separately according to group size. Detection was assumed to be certain out to 10 metres. For the remaining distances, the predicted Y values are divided by Y_{10} (the predicted value of Y at Dist = 10) to obtain g. A Horvitz-Thompson type estimator was then used to estimate abundance, which allows the detection probability (g) to be applied to each observation according to the covariates for that observation, as follows:-

$$\lambda = \frac{N}{A} = \frac{1}{A}\sum_{j=1}^{N_C}\frac{m_j}{g_j},$$

where m_j is the size of cluster j, g_j is the probability of detecting cluster j, N_C is the number of clusters and A is the search area. The estimates are determined sample by sample and then averaged over locations, dates, and sites to obtain an average bird density. Abundance was estimated using density and size of each site. Note that the constraints g>0.1 (all clusters) and $g\geq 0.5$ (clusters≥50) were used to reduce variability in the Horvitz-Thompson estimates, and sightings were included up to 300 metres from the observer when estimating area sampled and density. This method uses Poisson generalised linear models to estimate the likelihood of detecting a flock of birds according to a range of variables, and then corrects each individual record accordingly. While the 'Distance' Program (Thomas et al.

2010) can be used where distances are recorded, the approach taken here is much more flexible in modelling the detection data. It also uses the estimated detection probabilities associated with each observation to obtain Horvitz-Thompson estimates of population size, rather than obtaining marginal estimates of detection probability, which are then converted to population estimates. That is, for this study, each flock of birds observed was corrected using a different probability of detection according to the distance it was observed from the point, which species it was, the size of the flock and its activity. Much of the bird data in this study could not be modelled using the 'Distance' program owing to the issue of sparse cells which leads to convergence and other numerical problems with the program. A further difference is that while the 'Distance' Program uses specific parametric models (with or without polynomial adjustment terms) this approach uses smoothing splines which are fitted as a random term in a mixed model and therefore the shape of the detection function is essentially nonparametric.

Estimating bird damage

Bird damage was estimated in wine grapes and cherry, apple and pear orchards using methods described in Tracey and Saunders (2010) with the following differences. For cherries the numbers of damaged and undamaged cherries were counted in three cherry clusters, selected from one of 16 randomly selected zones in each selected tree. Trees from each block were systematically selected, with outside stratums sampled more intensively. To assess different types of damage, sampling occurred at various stage of flower, bud and fruit development. In apple and pear orchards the numbers of damaged and undamaged fruits were counted from one of six randomly allocated zones on each selected tree, with trees from each block systematically selected. To account for damage caused early in the season (especially by Sulphur-crested cockatoos, *Cacatua galerita*), old fruit on the ground under the tree was inspected for signs of damage. Cockatoo damage is distinct from damage by other species as they split the fruit in half to retrieve the seeds in the core.

Between 2000 and 2008, approximately 39,000 bunches of grapes, 31,000 apples and pears, and 27,000 cherries were sampled for bird damage in 326 blocks and 49 sites.

Experiments to reduce bird damage

Using the methods previously described, a field study was conducted across 101 orchards and vineyards over seven years (2000, 2001, 2004 – 2008) to test the efficacy of netting, shooting and scaring treatments in reducing bird damage. Generalised linear mixed model analyses were used to test for the effects of these treatments when either individually or in combination, on percent bird damage across 185 property year records. Included in the model as fixed effects, to account for variation, were crop type (grapes, apples, pears, cherries), square root of property size and treatment expenditure per hectare (where available, and allowing extra variation across results where not available). Random effects in the model were property (101), year (7), interactions between treatment and property and treatment and year, and finally random error. Stepwise regression was performed to remove non-significant terms (>0.05). LSD ranking across treatment levels was used to show significant differences. Generalised linear mixed model analyses were also used for netting, shooting and scaring separately, to test the effects of net type (extruded, black and green; drape-over, single, double, triple and quad; and permanent), the number of birds shot (shooting model) and compare scaring treatments (gas guns, electronic devices and visual deterrents), including interaction terms and significant fixed and random effects identified in the initial analysis. Generalised linear mixed models were fitted using the package *asreml* (Butler et al., 2009) under R (R Core Team, 2013). Wald tests were used to examine fixed effects and non-marginal terms not significant at the $P = 0.05$ level were dropped sequentially.

Expenditure per hectare for treatments included all anticipated direct up-front and on-going costs associated with the technique including labour ($AUS18.26/hour, Tasmanian Industrial Commission, 2006). Machine and vehicle hire and running costs (maintenance, fuel, labour) were included where appropriate. Up-front capital costs of netting as provided by the manufacturer (single drape-over $4356/ha, double row drape-over $4860/ha, triple row drape-over $4212/ha, side nets $3200/ha, extruded netting $2000), and were depreciated over 7 years. Netting machines were depreciated over 25 years, their life expectancy. Other costs included post caps for netting, labour in repairing and removing nets for spraying, and tying nets down where applicable. However, indirect costs associated with netting such as any changes to orchard management, increased pruning or spraying, removal of prunings from between rows etc. were not included. For shooting,

vehicle running costs were based on 60 cents per kilometre for a Ute or 4WD and 20 cents per kilometre for motorbikes. Ammunition was estimated to cost 38 cents per shell (7 ½ g shot).

Detailed information was collected to determine the ongoing and up-front costs of shooting, including number of shooters, time and duration of shooting activities, the number of shots fired, cost of ammunition, cost and type of firearm, number of birds shot, bird species, costs of vehicle (if applicable), location and habitat.

Scaring treatments involved the use of LPG and acetylene gas guns; electronic noise generating devices, using radar or automatic timing systems and predator, distress and alarm calls and artificial sounds; and visual deterrents, including kites with hawk silhouettes, reflective pyramids and CDs.

Experiments to reduce bird abundance

The relationship between bird density and damage for nil treatments (n=61 vineyards and orchards) was examined using a linear model with transformations of the data in *Program R* (R Core Team, 2013). A global test was also used to examine the linear model assumptions using the *gvlma package* (Pena and Slate 2006).

Short-term effects of nest removal (estimated using systematic nest searches and breeding data) and shooting on the abundance of starlings; and the effects of trapping on the abundance of starlings, blackbirds, silvereyes and sparrows (*Passer domesticus*) in fruit crops were investigated. Consumption of novel food types and estimated proportions of birds feeding were also used to evaluate the feasibility of poisoning for starlings and corvids.

Nest searches

Systematic searches and breeding behaviour of starlings in vineyards and orchards was used to investigate the efficacy of nest removal in reducing local populations of starlings. Searches of all potential starling nest sites were conducted in 9 vineyards in one breeding season and 1 vineyard over four breeding seasons, commencing in September. Subsequent searches and checking of nests occurred at least every 15 days, from September until February. All fence posts, buildings and accessible limbs and branches of every tree on

each site were systematically searched, and hollows individually marked and numbered. Measurements taken included entrance diameter, cavity depth, internal cavity diameter, hole orientation, circumference of tree below nest hole, tree species, tree height, diameter at breast height, and tree position. Investigation of nests was aided by a small torch and a borescope. Searches of native and introduced vegetation first occurred from the ground, then with binoculars, and then by climbing into each tree to search individual limbs with the aid of a 5 metre ladder. Where branches were inaccessible (>10 metres) searching was restricted to the use of binoculars from the ground, or from other accessible parts of the tree. Labour ($AUS18.26 per hour) was recorded during searches and for checking and marking of natural nest hollows.

To estimate the accuracy of nest searches in detecting starling nests more intensive observations (3-20 person days/week) were carried out on one site in two seasons. Frequent observations occurred during other detailed studies of starling behaviour, where nest use, flock size, habitat use, movements (banded and radio-tagged birds), and encounters between starlings and native species were regularly recorded through-out the day from September to February. During these studies one to four people frequently undertook observations (3-20 person days/week) in vineyards and the surrounding areas, and any un-marked starling nests were recorded.

Trapping

To evaluate the efficacy of trapping to reduce pest bird abundance, 76 trapping events were conducted for starlings, sparrows, blackbirds and silvereyes. A trapping event being a period of trapping during a season (a three month period: Summer, Autumn, Winter or Spring) over eight years (2000 - 2008) with trapping occurring in 21 season years on 22 sites. Birds were captured using mist nets, modified Australian crow (MAC) traps, Tidemann two-stage traps, and smaller walk-in cage traps using methods of Lukins and Tracey (2013). During Spring and Summer (September- February, 6 season years on 7 sites) adult starlings were also captured in nest box traps, and by hand from natural nest sites and nest boxes with the aid of a plywood panel with a drop-down door, which was transportable and was placed inside boxes or outside natural entrances to capture starlings where they were known to be nesting. Nest boxes (40 x 25 x 25 mm) were constructed from 10mm plywood and painted for weather protection. Perches of 12mm dowel were attached below the 80mm diameter entrances, and a hinged roof allowed investigation of

nesting activity. Boxes were fastened to trees using tech screws and holes were drilled in the base of the boxes for drainage purposes. All boxes were attached with a southern facing orientation (Pell and Tidemann, 1997). Labour ($AUS18.26 per hour) was recorded for checking and marking of natural nest hollows, free-feeding and checking of cage traps, nest boxes and nest box traps. Percent and proportional reduction in pest bird populations were estimated separately for starlings, blackbirds, silvereyes and sparrows using the number of birds captured and abundance estimated for each site and season (as described). Introduced species (starlings, blackbirds, sparrows) were removed after capture, while silvereyes were banded, to account for recaptures (5.8%, n=51), prior to release. Many non-target species were captured, particularly when using mist nets, including Australian magpie (*Cracticus tibicen*), Australian raven (*Corvus coronoides*) , Australian reed warbler (*Acrocephalus australis*), brown falcon (*Falco berigora*), brown goshawk (*Accipiter fasciatus*), brown thornbill (*Acanthiza pusilla*), buff-rumped thornbill (*Acanthiza reguloides*), Cunningham's skink (*Egernia cunninghami*), crested pigeon (*Ocyphaps lophotes*), crimson rosella (*Platycercus elegans*), diamond firetail (*Stagonopleura guttata*), Eastern blue-tongue lizard (*Tiliqua scincoides*), eastern rosella (*Platycercus eximius*), eastern spinebill (*Acanthorhynchus tenuirostris*), golden whistler (*Pachycephala pectoralis*), grey butcherbird (*Cracticus torquatus*), grey fantail (*Rhipidura albiscapa*), grey shrike thrush (*Colluricincla harmonica*), laughing kookaburra (*Dacelo novaeguineae*), magpie lark (*Grallina cyanoleuca*), Pacific black duck (*Anas superciliosa*), pied currawong (*Strepera graculina*), red browed finch (*Neochmia temporalis*), red wattlebird (*Anthochaera carunculata*), red-rumped parrot (*Psephotus haematonotus*), speckled warbler (*Chthonicola sagittatus*), spiny cheeked honeyeater (*Acanthagenys rufogularis*), spotted pardalote (*Pardalotus punctatus*), striated pardalote (*Pardalotus striatus*), striated thornbill (*Acanthiza lineata*), superb fairy wren (*Malurus cyaneus*), white browed scrub wren (*Sericornis frontalis*), white plumed honeyeater (*Lichenostomus penicillatus*), white-browed babbler (*Pomatostomus superciliosus*), willie wagtail (*Rhipidura leucophrys*), yellow faced honeyeater (*Lichenostomus chrysops*), yellow rumped thornbill (*Acanthiza chrysorrhoa*), and yellow thornbill (*Acanthiza nana*). These were released at the capture location.

Generalised linear mixed model analyses were used to examine the effects of species captured (4), season (4), trap type (4), trap days (12-749), size of site (2- 500.9 ha) and

interactions on the proportion of birds captured (n=76 trapping events). A logit (log (p/1-p)) transformation was used for the proportion of birds captured (p) to remove heterogeneity of variance. generalised linear mixed models were fitted using the package *asreml* (Butler et al., 2009) under R (R Core Team, 2013). Stepwise regression was performed to remove non-significant terms (P>0.05). Wald tests were used to examine fixed effects and non-marginal terms not significant at the P = 0.05 level were dropped sequentially. A t-test with equal variances was used to compare the large cage traps (MAC, Tidemann two-stage) for starlings.

Bait consumption and preferences

For starlings, bait consumption and preferences were estimated for six bait types: dog biscuits; poultry layer pellets; sultanas; table grapes (cage trial only); whole wheat and bread and dripping. Bread was cut into pieces 8-12mm square and then dipped in hot dripping and rolled in flour. All bait materials, with the exception of bread and grapes, were coated with a small quantity of vegetable oil. The oil was added to simulate the condition of material containing a toxin – vegetable oil is used to bind the chemical to the bait material when poison baits are prepared.

In cage trials, twenty starlings, captured in the wild with two-stage cage traps, were placed in 5 cages (800 x 800 x 900 (h) mm from 1.2mm weld mesh 25mm x 25mm); with two males and two females per cage. Six types of bait media, provided in individual trays, were replenished and weighed daily for 16 days to determine the quantities of bait consumed per day.

The justification for use of animals, the number of animals used, assessment of the animal's well being, free-feeding and capture, handling, housing, care were approved and monitored by the NSW Animal Care and Ethics Committee. Animal Research Authorities (ORA 99/ 014; ORA 01/003; ORA 05004; ORA 04008) were received, as required under the *Animal Research Act 1985*. Standard operating procedures (Lowe 1989; Sharp and Saunders 2004a, b, c; Lukins and Tracey 2013) were followed for pre-feeding, timing, trap operation, housing, and handling, care and euthanasia of birds and welfare, safety and legal requirements were met.

Experienced bird handlers conducted all procedures. Standard measurements of weight, head-body (HB), bill (BK) and tarsus with foot (TZ) lengths were taken as described in the

Australian Bird Banders Manual (Lowe 1989). Initial trapping was not conducted in excessively windy or hot periods or during rain, and traps were continuously monitored and checked during trapping periods. Maximum time between checking of traps varied with capture method (Lukins and Tracey 2013) from 20 minutes (mist net) to two days (large cage traps). Transport of birds from the field occurred using a holding cage or soft calico bag. Suitable shade and an adequate number of perches were available in holding cages, and daily servicing took place to ensure clean water and food, and to monitor bird health. At the completion of the study birds were removed from the cage by hand or a hand held net and euthanased by cervical dislocation or carbon dioxide (CO_2).

In field trials, continuous observations of birds visiting non-toxic bait trays were conducted at four sites over a 25 day period. At each site galvanised steel trays (measuring 1000 x 500 x 25 mm), or later round black plastic trays (360 mm in diameter) were laid out and held in place by pegs. A group of five trays were placed on the ground spaced 0.5 m apart. At each site, groups of trays were placed in two or three separate locations, 50-400 m apart with the same locations chosen throughout the trial. Any pasture surrounding the trays (within approximately 1 m) was trimmed using a brushcutter.

Bait substrates were weighed and placed in a mound on the feed trays in the following order; bread and dripping, dog biscuits, sultanas, pellets and wheat. At the end of each observation, bait substrates that had been visited by birds were again weighed and change in mass was recorded. Bait material was topped up or replaced to the required weight (500 g ± 1%) as necessary. Observations of 1 hour duration from a bird hide were conducted through-out the day (6:00 – 18:00) with a total of 318 hours. All feeding and non-feeding visitations by any species to each bait type were recorded, with number of birds and length or feeding recorded.

In the field trials, standardised counts were also conducted hourly during daylight on each site, and abundance estimated. The proportion of birds feeding was estimated using the maximum number of birds observed feeding from trays after 20 days of free-feeding, and estimated total bird abundance.

The effects of bait type, abundance, number of days of feeding, site, time of day on the number of birds visiting feeding trays were examined using generalised linear mixed models in Program R (as described above). Analyses were conducted separately for each of

the 3 species; starlings, corvids (Australian and little ravens) and magpies. The Poisson distribution was used in all analyses, and a dispersion argument included for the starling model, as this data was over-dispersed.

4.3 Results

Bird species

One-hundred and forty bird species were observed in vineyards and orchards during the study (*n*= 2,466 point locations, 101 sites), including nine introduced species. Of these 29 native and 7 introduced species are known to consume fruit (Tracey et al., 2007). The main pest species were starlings (70.4%, mean density 419± 112 starlings/ km^2), eastern rosellas (*Platycercus eximius*) (4.7%), pied currawongs (*Strepera graculina*) (4.3%), corvids (Australian ravens, *Corvus coronoides* and little ravens, *C. mellori*) (3.9%), silvereyes (3.5%), sulphur-crested cockatoos (3.4%), red wattlebirds (*Anthochaera carunculata*) (3.1%), black-faced cuckoo-shrikes (*Coracina novaehollandiae*) (2.3%), crimson rosellas (*Platycercus elegans*) (2.1%), noisy miners (*Manorina melanocephala*) (1.9%), yellow-faced honeyeaters (*Lichenostomus chrysops*) (1.6%), noisy friarbirds (*Philemon corniculatus*) (0.4%) and blackbirds (0.2%).

Experiments to reduce bird damage

The generalised linear mixed model analyses indicated a significant effect for property size (Wald statistic: $F(1, 138) = 4.901$, $P = 0.028$) and a significant Net x Shooting interaction (Wald statistic: $F(1, 78) = 6.716$, $P = 0.011$). Bird damage was significantly lower on netting (10.7± 2.8%) and shooting (20.5± 3.8%) sites than nil treatments (33.2± 5.6%); and netting was more effective than shooting. The mean cost of netting and shooting treatments was \$1,903±327/ha/site and \$538±199 /ha/site respectively. Scaring and control effort (measured by treatment expenditure per hectare) had no effect on bird damage and there were no significant differences in damage between crops (apples, pears, cherries or grapes). There were no significant differences between net types (permanent, drape over, extruded) and the number of birds shot had no effect on bird damage. Lower damage was reported on sites with gas guns compared to those with electronic devices and visual deterrents (Wald statistic: $F(1,74)=7.158$, P=0.028).

Table 4.1. Mean bird damage (%) across all crops of average property size (12.3 ha), associated standard errors and an LSD ranking across treatment levels where there were significant differences.

Net	Shooting	Mean Damage (%)	Standard Error (%)	LSD Rank
No	No	33.2	5.6	c
No	Yes	20.5	3.8	b
Yes	No	10.7	2.8	a
Yes	Yes	15.2	4.0	ab

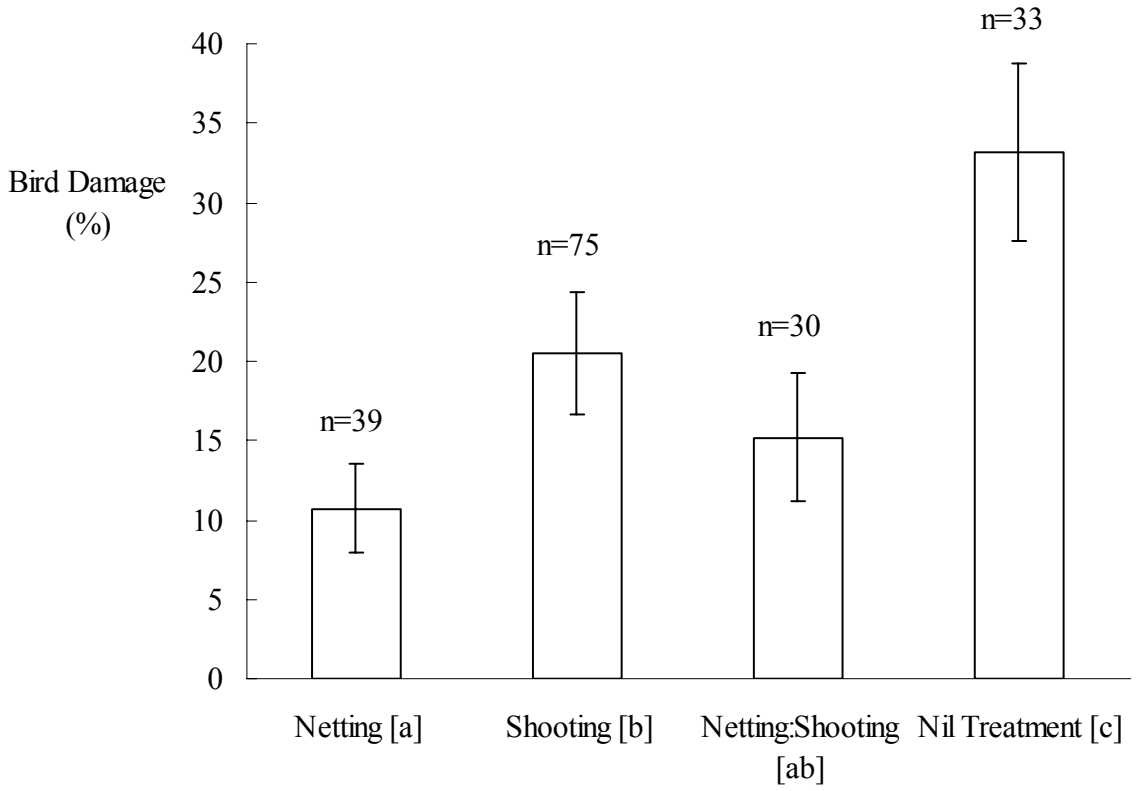


Figure 4.1. Bird damage (%, standard error) in vineyards and orchards of average property size (12.3 ha) under shooting, netting and nil treatments. LSD rank [ ] shows significant differences across treatment levels.

Experiments to reduce bird abundance

A significant positive linear relationship ($y = 0.0003x + 0.8053$) was found between bird density (x; birds per km^2) and the $\log_{10}$ of bird damage (y) (Figure 4.2; $F_{1,60} = 8.775$, $P = 0.00437$; *adjusted* $r^2 = 0.1131$), with the linear model assumptions assessed as suitable (Table 4.2). The low *adjusted* r^2 value and therefore predictive value of the linear relationship is likely a result of the high variability in damage when bird density is low. This indicates that high levels of damage were still evident in vineyards and orchards with low bird density, but that a high density of birds (>1000 birds per km^2) is likely to result in significantly higher levels of damage. A $\log_{10}$ transformation of bird density reduced the scatter of points, but did not increase the predictive value (*adjusted* $r^2 = 0.10$).

Table 4.2. Assessment of the linear model assumptions using the global test (Pena and Slate 2006; Kabacoff 2001). Level of significance = 0.05.

	Value	*p-value*	*Decision*
Global Stat	4.71748	0.3175	Assumptions acceptable.
Skewness	1.31872	0.2508	Assumptions acceptable.
Kurtosis	0.01969	0.8884	Assumptions acceptable.
Link Function	2.56074	0.1095	Assumptions acceptable.
Heteroscedasticity	0.81832	0.3657	Assumptions acceptable.

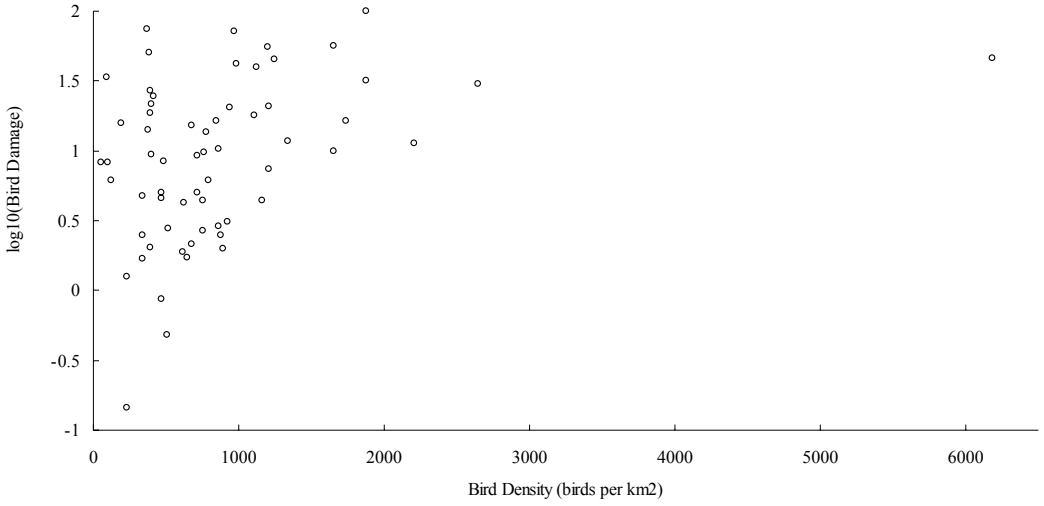


Figure 4.2. Relationship between bird density (birds per km^2) and $\log_{10}$ (bird damage) in vineyards and orchards of the Orange Region, New South Wales.

Nest searches

Six hundred and twenty-two hollows were marked on ten sites (mean 62.2 ± 26.9 per site); 78.1 ± 6.9% of these were accessible and 17.0 ± 7.8% had starling nests. In addition, intensive observations on one site over two seasons indicated that 4.2 ± 2.7% of starling nests were missed during total searches. Taking these into consideration, 81.9 ± 8.7% (*n*=13 breeding season sites) of starling broods were accessible for removal. Starlings laid 4.16 ± 0.25 (range 1-6, *n*=85) eggs per nest, during September, October and November.

Trapping

The interaction between species and trap type was significant in the final trapping model (Wald statistic: $F(1,202)=42.2$, $P<0.001$), with walk-in traps more effective than other capture methods for blackbirds, mist nets for silvereyes, and large cage traps for starlings (Figure 4.3). Season, trap days and size of site had no effect on the proportion of birds captured. Capture rates of starlings for the MAC (mean 3.54 ± 0.98 birds per trap day) and Tidemann two stage traps (mean 3.61 ± 0.83 birds per trap day) were not significantly different (*t* statistic = 0.056, *n*=14 site season years, 334 trap days, *P*=0.477).

For starlings, over 60% (46 of 76) of trapping events resulted in zero captures, 9.2% (7 of 67) resulting in reductions of over 30%; and trapping reduced the local population of starlings by over 90% on 2 occasions. Mean costs per bird were $14.83 overall (1789 birds captured); $14.76 (1292 captured) for starlings in large cage traps, $32.66 (123 captured) for adult starlings captured in nests or nest box traps, $21.20 (134 captured) for blackbirds in walk-in traps, and $11.82 (51 captured) for silvereyes in mist nets. The majority of starlings captured in nest boxes were male (83%, *n*=24).

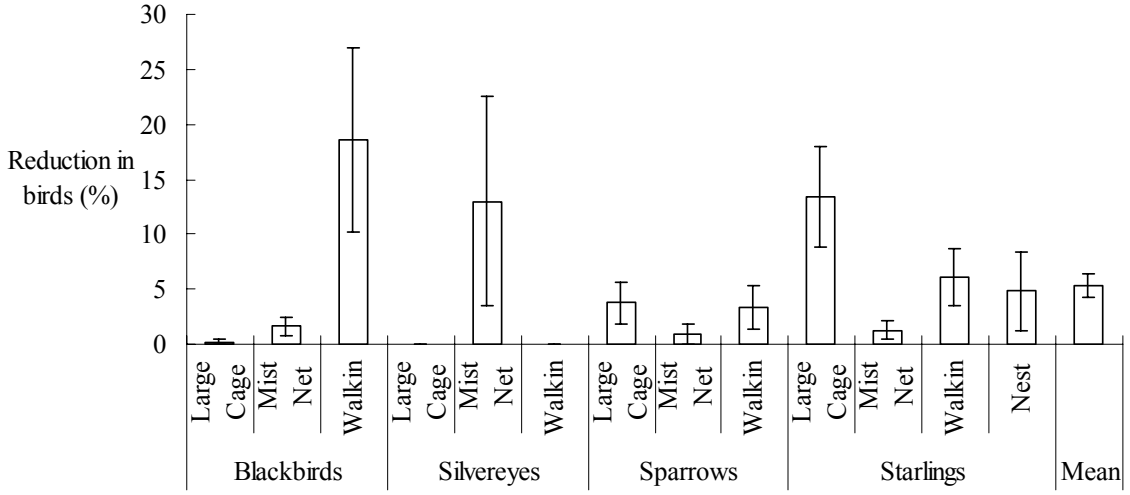


Figure 4.3. Reduction (%, standard error) in pest birds by species and capture method from trapping in vineyards and orchards of the Orange Region New South Wales.

Bait consumption and preferences

Sultanas (6.3±1.6 grams/starling/day), dog biscuits (6.0±2.7) and bread and dripping (4.8±0.8) were preferred food items for starlings during the cage trial, with approximately equal proportions consumed. Table grapes (1.26± 1.2) and poultry pellets (1.17± 0.96) were occasionally consumed, and wheat was avoided (0.32±0.18). The overall mean quantity consumed per starling per day was 19.90 ± 1.72 grams. Slight changes in diet were observed over the cage trial period, with starlings switching preferences from dog biscuits to sultanas (A power function for the proportion of bait media consumed by starlings, y versus the time in activity x, $y = 0.17\ x^{0.30}$, $r^2 = 0.66$ for sultanas, and a log function, $y = -0.12 \ln(x) + 0.52$, $r^2 = 0.71$, for dog biscuits showed a good fit to the data). Consumption of other food types was constant during this period.

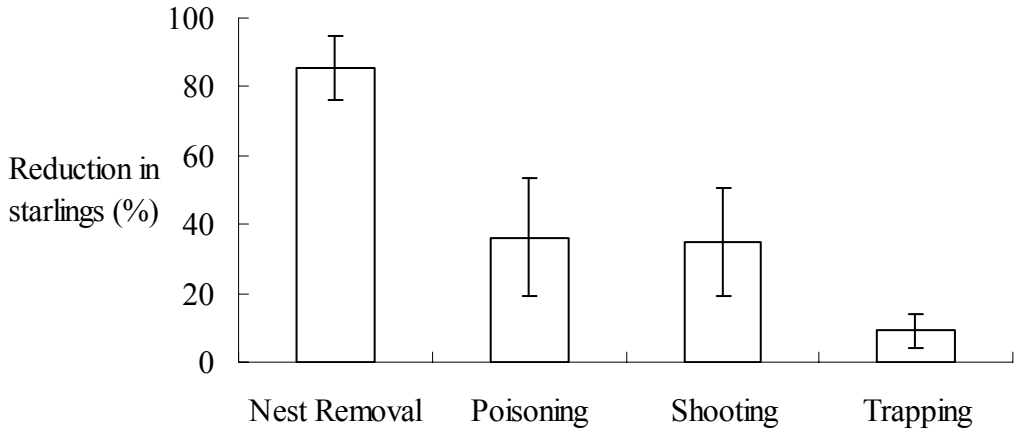
During bait field trials, starlings were most abundant and the most common species to visit (60%, n=1609) and feed from (53%, n=1564) bait trays. Magpies (23%, n=1564), corvids (12%, n=1564), magpie larks (5%, n=1564), and white-winged choughs (4%, n=1564), sulphur-crested cockatoos, crested pigeons and pied butcherbirds were the only other species observed feeding from trays.

For starlings, bait type, free-feed days and abundance were significant ($P<0.0001$) in the final model. Bread and dripping (1.95± 1.4 starlings/hr), sultanas (1.68± 0.97 starlings/hr), and dog biscuits (1.95± 1.4 starlings/hr) were preferred, while poultry pellets (0.02± 0.02 starlings/hr) and wheat (0 starlings/hr) were avoided. The percentage of birds feeding and bait consumption increased after 20 days (Before 2.8±1.8% vs After 36.0±9.5%).

For corvids, bait type and free-feed days were significant ($P<0.0001$) in the final model. Corvids displayed a clear preference for bread and dripping over other bait media presented, and visits increased gradually with days of free-feeding (Poisson: size of effect for free-feed days =0.0213). For magpies, bait type and property were significant ($P<0.0001$) in the final model, with an obvious preference for bread and dripping. A higher number of starlings visited trays (4.46 ± 0.85 per hour, n=194 observations), than corvids (1.08 ± 0.85) and magpies (2.01 ± 0.85).

Systematic searches indicate that nest removal would be highly effective (81.9 ± 8.7%) in reducing starling reproduction. Poisoning and shooting were equally effective and trapping least effective in reducing starling abundance (Figure 4.4a). Nest removal (estimated from systematic searches), poisoning and trapping were equally efficient ($/bird) and shooting most expensive ($/bird) for starlings (Figure 4.4b).

(a)



(b)

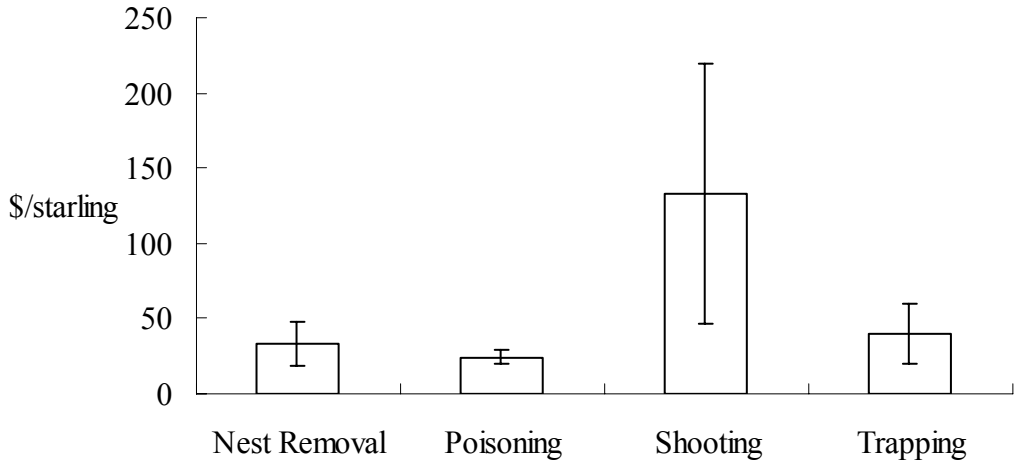


Figure 4.4. Effectiveness (a) and efficiency (b) of lethal controls for reducing starling abundance (poisoning, shooting, trapping) and reproduction (nest removal, estimated from systematic searches) in vineyards and orchards.

4.4 Discussion

Experiments to reduce bird damage

Netting is often assumed to offer complete protection from bird damage (Stucky, 1974; Foster, 1979; Fuller-Perrine and Tobin, 1993; Tracey and Vere, 2007). However, whenever measured, damage is commonly reported under nets and was often observed in my study (mean 10.7± 2.8 %) with damage as high as 56.4± 4.8%. Birds breached nets through holes or gaps in netting, created as a result of catching on posts and wire, wind or in-complete installation. They also consumed fruit through nets, while smaller species entered through nets with large mesh size. For example, pied currawongs were observed consuming fruit through nets, including finer mesh and silvereyes and yellow-faced honeyeaters entered through and were regularly observed under netting. Somers and Morris (2002) also reported high levels of damage through nets, even when mesh size was small. In some cases results are excluded from experimental studies as a consequence of the difficulties with netting installation (Berge et al., 2007). However, even with these difficulties netting was still found to be the most effective method in reducing damage.

Despite its widespread use, there have been few evaluations of the efficacy of shooting in reducing bird damage. In my study shooting (20.5± 3.8%) was not as effective as netting (10.7± 2.8%), but less than one-third of the cost ($538 vs $1,903/ha/property) and reduced damage compared to nil treatments (33.2± 5.6%) by 13%. In the only other study (Murton and Jones, 1973) intensive shooting of wood pigeons did not significantly reduce bird damage to brassica crops. In their study, shooting was conducted to reduce the overall population of wood-pigeons in a large area, rather than shooting to scare birds from individual crops. The benefits of shooting reported in my study, are likely to be a result of scaring birds from the crop, as the number of birds shot was unrelated to the extent of damage caused, and the numbers shot were low in relation to the pest bird population size (35.0±7.9%).

Scaring with noise- generating and visual deterrents had no effect on bird damage indicating that although birds may respond initially to scarers, they quickly acclimatise to new stimuli that do not pose a physical threat, which is supported elsewhere (Bomford, 1990a). Additional analyses of scaring indicated that gas guns were more effective than

electronic devices and visual deterrents. This is despite the use of radar-activated electronic systems and those with distress or alarm calls, which have been shown to decrease bird activity (Ronconi and St Clair, 2006: radar-activated for waterfowl on tailings ponds; Ribot et al., 2011: alarm calls for rosellas in apple orchards).

Control effort was not the same on all sites, leading to potential bias if not considered, for example, growers experiencing high levels of damage may be more likely to apply increased controls. This would result in underestimation of treatment effectiveness. However, we found control effort, estimated by the treatment expenditure per hectare, had no effect on bird damage. Similarly Murton and Jones (1973) found that the amount of damage was not correlated with the amount spent on crop protection.

The size of the crop was important in the current study, with increased damage on smaller plantings, which is commonly observed (e.g. grackle damage to grapefruit Johnson et al., 1989). With equivalent bird abundance the proportion of fruit consumed by birds is expected to be less on larger vineyards and orchards, as more fruit is available. This effect may be exacerbated by birds' preference for the outer edges, which is common in bird-crop conflicts (Johnson et al., 1989; Somers and Morris, 2002; Tracey and Saunders, 2010).

Experiments to reduce bird abundance

Measures to reduce pest bird abundance in the long-term or over large areas are rarely effective (Feare, 1991), and not desirable for native species (29 of 36 pest species in my study). However, short term reductions in pests may have benefits in reducing damage (Palmer, 1972; Ward, 1979) or in preventing pest establishment into new areas (Woolnough and Parry, 2007).

A positive linear relationship between bird density and damage (log_{10}) indicates that a reduction in bird density would be beneficial in reducing bird damage to orchards and vineyards. However, there was high variability in damage when bird density was low with poor predictive value. This is true also in other pest damage functions (e.g. wild dogs, Fleming et al. In Press) and emphasises the importance of monitoring damage directly where possible (Tracey et al. 2007), and targeting pests where damage is high, rather than broadscale control of pests wherever they occur.

Results indicate that nest removal would be highly effective (82%) and relatively efficient ($30 / starling) in reducing starling reproduction. However, compensatory breeding (Newton, 1994) following nest removal and immigration was not estimated, which is likely to limit effectiveness, and capture of adults at nests and with nest box traps had limited success (5.34 ± 1.1% reduction). Starlings with unsuccessful clutches are likely to lay replacement eggs, but these are less successful (Feare, 1984). However, controls of adults during nesting, and the timing of nest removal are likely to affect these compensatory responses i.e. removing nests and euthanising juveniles just prior to fledging would reduce the likelihood of starlings raising a replacement brood.

Starlings have high breeding potential (2 broods per year, 4.6 per pair, my study), high rates of increase (Feare 1984), and high rates of natural mortality, particularly in the first year (Feare 1984). Hence, control of breeding in late winter and spring is likely to result in longer term reductions in populations than control at other times of the year. Conversely, control during the ripening period (late summer, autumn), when large numbers of juveniles are present, may have immediate benefit (Conover and Dolbeer, 2007), but is unlikely to result in reduced recruitment in the following breeding season. While my results indicate high efficacy of nest removal, further investigation is needed to evaluate the longer-term effects on starling populations.

The effectiveness of different traps varied with species. Walk-in traps were more effective for blackbirds, which is consistent with their foraging and social behaviour. Blackbirds forage for food in small numbers amongst shrubs or dense vegetation. Conversely starlings forage in large flocks preferring open agricultural areas, hence the large cage traps were more effective than other capture methods. Capture rates between MAC and Tidemann two stage traps were similar for starlings, which differs from Campbell et al., (2012) who found Tidemann two stage traps to be 1.5 times more effective.

Mist nets were more effective than cage traps for silvereyes, which were captured along windrows or whilst flying into vineyards from adjacent habitat. Silvereyes were not attracted to cage traps or food presented. Plesser et al., (1983) reported the capture of 2754 sparrows over 10 days in a vineyard, by placing 57 metres of net 2m from the outside vineyard row. They reported the elimination of damage to the vineyard as a result, and attributed their success to the relatively small area (1.4 out of 10ha) that had ripe grapes

during the trial. However, the use of mist nets as a continuous barrier to protect fruit would be time consuming and impractical for large vineyards and orchards. The checking of mist nets every 20 minutes is required to ensure birds are removed before they are heat stressed or attacked by predators (Lowe, 1989; Lukins and Tracey, 2013), and restrictions apply on their use in most countries. These nets are not target specific (with 33 non-target species captured during our study), and many native species may be present (Conover, 1982; 140 species observed in our study). Mist nets, large cage traps and walkin traps were equally effective for sparrows. These species were caught in close proximity to buildings and sheds, which is consistent with their local movements and feeding behaviour (Plesser et al., 1983; Higgins et al., 2006).

The capture of adult starlings with nest box traps and at nest hollows was less effective than cage trapping. However, this capture method in combination with nest removal, could be used to reduce compensatory breeding; and target populations when their abundance is lowest. The majority of the starlings captured in nest box traps were male (83%, n=24), which is likely a consequence of their role in finding suitable nest sites and building nests.

Overall, trapping was difficult and costly. Greater success has been reported elsewhere (Dolbeer, 1989). MAC traps, in particular, are commonly used by orchardists for corvids (Moran, 1991), blackbirds (Moran et al., 2004), starlings and house finches, where large numbers are captured. For example, Elliot (1964) reported the capture of 110,000 starlings to reduce damage to cherries in Washington; Larsen and Mott (1970) reported the capture of 3500 house finches in blueberries; and Palmer (1972) used trapping and poisoning to reduce finch damage to figs in California. However, independent measures of abundance or damage were not used to evaluate effectiveness. Conover and Dolbeer (2007) reported an immediate 80% (>500 before vs <100 afterward) reduction in starlings foraging in a blueberry orchard using MAC traps. Conover and Dolbeer's (2007) study however, is an example of success in one orchard. Similarly, in my study trapping was successful in reducing the local starling population by 90% on two sites, but overall effectiveness was low. This may emphasise the need to consider publication bias in future (Section 2.1), where authors maybe more likely to report positive results in isolated cases rather than considering efficacy across the range of situations where control methods are applied.

Trapping for starlings is usually conducted during summer when large flocks of juveniles are present, which also coincides with fruit ripening. Trapping success for starlings is often attributed to the naivety of juvenile birds, which comprise the majority of captures (Conover and Dolbeer 2007). However, season did not significantly affect the proportion of birds captured in my study. Presence and number of lure birds (3: Campbell et al., 2012; 4-6: my study; 20-40 Conover and Dolbeer 2007) can also affect trap success. Conover and Dolbeer (2007) reported that MAC traps were only effective when 10-20 starlings were used. Food availability and preferences for food presented is also likely to affect trap success.

Grain-based pellets using either poultry layer mash (Johnson and Glahn, 1994), corn (Knittle et al., 1980) or rice (Linz et al., 1995) are the recommended and most commonly used bait media for starlings and American blackbirds (red-winged blackbirds, *Agelaius phoeniceus*, yellow-headed blackbirds, *Xanthocephalus xanthocephalus*, and grackles, *Quiscalus quiscula*). While poultry pellets were consumed by starlings in my study, bread and dripping, sultanas and dog biscuits were consumed in preference in both field and/or cage trials. Sultanas were the most target-specific for starlings, with corvids and magpies commonly consuming bread and dripping and occasionally dog biscuits. Familiarity with food is important in trapping success, for example, poultry feed is preferred at feedlots (West and Besser, 1976), grain in grain growing and storage areas, fruit in vineyards, almonds in nut orchards (Sinclair, 1998), and bread or dog biscuits in residential areas (Nelson, 1994; Lukins and Tracey, 2013).

Diluting poison bait with untreated feed has been recommended previously for reducing non-target impacts, for example 1:25 (Linz et al., 2002), or up to 1:200 is recommended (West et al., 1967). However starlings in my study were less abundant (e.g. mean 260 starlings per property versus 77,000, Knittle et al., 1980), required longer periods of free-feeding (23 days vs 1-4 days Johnson and Glahn, 1994) compared with other studies (West et al., 1967; Linz et al., 2002). My results indicate that high dilution rates may reduce effectiveness for starlings, while still exposing ravens and magpies, which fed more consistently and for longer periods than starlings. Although not assessed here in relation to non-target abundance and risk, observations suggest magpies, corvids and magpie larks would be the main non-target species from a starling trapping or baiting program. Results also indicate that a small number of white-winged choughs, sulphur-crested cockatoos,

crested pigeons and pied butcherbirds may also be exposed. Other non-target species may be at risk in other areas for example along watercourses or near dams. Coots (*Fulica atra*), mallards (*Anas platyrhynchos*), Pacific black ducks (*Anas superciliosa*), wood ducks (*Chenonetta jubata*), and grey teal (*Anas gracilis*) are easily attracted to grain or bread, and anatids are susceptible to the DRC-1339 (Eisemann et al., 2003), as well as other avicides.

Murton and Jones (1973) showed that the likelihood of bait acceptance increased with bait density, which could have limited feeding activity and explain the length of free-feeding needed in my study, where relatively small amounts of feed were presented.

Knittle et al., (1980) suggest that bait consumption estimated using cage and field trials is the most reliable method of estimating overall effectiveness of baiting compared to other methods, including using change in population estimates and/ or numbers of dead birds recovered during systematic searches. My results indicate a mean reduction of 36.0±9.5% could have been achieved after 20 days of free-feeding. This is likely to be conservative as individual birds were not able to be identified, and the maximum number of birds (not total birds) observed feeding from trays was used. While this period of free-feeding is longer than recommended in other studies (1-4 days, Johnson and Glahn, 1994), differences in the familiarity with food types, varying feeding rates, and therefore efficacy are expected in different locations.

Increased baiting efficacy and higher feeding rates by starlings were recorded by Knittle et al., (1980), where there larger congregations of birds were present (77,000), larger amounts of food was presented (23 kg per site), and where baiting occurred during winter, when alternative foods were likely to be limited. In comparison, the current trial sites were located in a productive agricultural area (mean annual rainfall: 920 mm), where their preferred foods, including ground invertebrates (Coleman, 1977) and fruit are regularly available and widely distributed. Greater success may be expected in drier conditions, where pasture growth is limited and alternative foods are lacking. In addition many of the smaller orchards and vineyards would occupy only a proportion of a starling's home range. Immigration of birds from surrounding areas is therefore likely, which could explain the limited effectiveness of lethal controls reported here.

Communication between birds is also likely to be important in finding food, which may increase with increased abundance. For example, birds find food more efficiently in a flock (Krebs et al., 1972), and large communal roosts may be important as 'information centres', particularly where food is unevenly distributed across the landscape (Ward and Zahavi, 1973).

Bread and dripping was the preferred bait type for corvids and magpies. These species often feed on carrion (Higgins et al., 2006); and meat baits, offal, small mammal carcases or eggs are regularly used to poison or capture them (Johnson, 1994; Lukins and Tracey, 2013). However, other non-target species are also likely to be attracted to meat baits. For example in my study wedge-tailed eagles (*Aquila audax*), little eagles (*Hieraaetus morphnoides*), whistling kites (*Haliastur sphenurus*), black kites (*Milvus migrans*), pied (*Cracticus nigrogularis*) and grey (*Cracticus torquatus*) butcherbirds, pied currawongs, grey shrike thrush (*Colluricincla harmonica*) (eggs), and mammals including feral pigs (*Sus scrofa*), dogs (*Canis lupus familiaris*), foxes (*Vulpes vulpes*), cats (*Felis catus*), and native (Muridae) and introduced (*Rattus rattus, Mus musculus*) rodents were present in vineyards and orchards and surrounding vegetation during the study. Other food types are also used for corvids and magpies, such as crumbed cheese, butter, corn, milo heads, watermelon, and poultry feed (Johnson, 1994; Nelson, 1994; Lukins and Tracey, 2013).

Results indicate that the use of poisons would be ineffective in vineyards and orchards. In addition, while the risks posed to non-target species was not assessed; many native species were present (131), with seven of these observed feeding from trays. Secondary risks of poisons would also need to be assessed prior to commencing a poisoning campaign, with at least seven birds of prey, five other bird species and six mammal species present that may be at risk by consuming sick and dying birds exposed to poisons.

Conclusions

Most studies of bird control measures have involved limited replication and/or failed to independently measure damage and pest bird abundance. Evaluations of bird management techniques require large sample sizes to account for the complexity of bird- crop systems and should incorporate, or attempt to control for, the variability in bird species, abundance and damage over time and between sites.

Despite their widespread use, results indicate that lethal methods used to control starlings, blackbirds, sparrows, silvereyes, and crows on vineyards and orchards have limited effectiveness in reducing pest populations in the short term. However, systematic searches indicate that nest removal would be highly effective at limiting reproduction and could have benefits for long-term management of starling populations. Results indicate also that shooting as a method to scare birds from the crop, rather than to control populations, is effective in reducing damage, while scaring with electronic devices and visual deterrents is ineffective.

Results indicate use of poisons for birds in vineyards and orchards may have limited effectiveness and is problematic due to likely but un-measured non-target affects including primary and secondary poisoning. In my study 131 native species were recorded in vineyards and orchards. Long term reductions in these populations are not feasible or desirable. However, feeding trials indicate that magpies, corvids and magpie larks may be the main non-target species at risk from a starling trapping or baiting program, and use of sultanas for starlings may reduce this risk.

Netting was the most effective method in reducing damage and has considerable advantages when large numbers of native species are present. However, damage under netting can be significant and should be considered before investing in nets over other methods. Selecting appropriate netting for the species involved, avoiding nets in direct contact with fruit and improved applications and maintenance of netting would improve effectiveness.

The effectiveness of non-lethal and lethal controls was highly variable between control methods and combinations, bird species, abundance, trap types, and food presented. Careful consideration of the species involved, their abundance, movements, and feeding preferences will improve the effectiveness and target specificity of controls for pest birds.

PART B: ENVIRONMENTAL IMPACTS

Preface

Part B considers environmental impacts of birds using introduced mallards on Lord Howe Island as a case study. In Chapter 5 I review the history of introduction and movements and provides current information on the abundance, distribution, activity, habitat use and breeding of Pacific black duck and mallard on Lord Howe Island. The extent of hybridisation between Pacific black duck and mallard is evaluated, and management options for mallards on the island are then considered. Pacific black duck and grey duck are both used as common names for *Anas superciliosa*. Pacific black duck has been used in this thesis.

5. Hybridisation between mallard and Pacific black duck on Lord Howe Island and management options

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Abstract

Introduced mallards (*Anas platyrhynchos*) occur on many islands of the South Pacific, where they hybridise with the resident Pacific black duck (*A. superciliosa*). In October 2007, we conducted systematic surveys of Lord Howe Island to estimate the abundance and distribution of Pacific black ducks, mallards, and their hybrids. Hybrids were common in areas of high public use, particularly where there was mown or grazed grass. Phenotypic characteristics suggest that mallards are now dominant and have supplanted the native Pacific black duck, with 81% of birds classified as mallard or mallard-like hybrids, 17% as intermediate hybrids and only 2% as Pacific black duck-like hybrids. No pure Pacific black duck were observed. These hybrids pose direct impacts to Pacific black ducks and

perceived but un-quantified indirect economic, social and environmental impacts to Lord Howe Island. A management program to remove mallards using trapping, shooting and opportunistic capture by hand was conducted in October 2007. Standardised indices of duck abundance before and after management indicates that the total population was reduced by 71.7%. If warranted, eradication of mallard and hybrids from Lord Howe Island is considered achievable with a program of education, monitoring, and continued control to prevent re-establishment.

5.1 Introduction

Pacific black duck (*A. superciliosa*) are widely distributed throughout the South Pacific (Marchant and Higgins, 1990). They favour fresh and brackish water and are uncommon in marine habitats, except during drought (Goodrick, 1979), or on oceanic islands (Horning and Horning, 1974; Norman, 1990). Movements on mainland Australia and New Zealand are associated with the availability of surface water (Roshier et al., 2001). Populations are sedentary near permanent water, and more dispersive in ephemeral wetlands and in times of drought (Frith, 1963, 1982). There are 3 subspecies of Pacific black duck (Amadon, 1943a; cf. Marchant and Higgins, 1990; Rhymer et al., 2004); the nominate *superciliosa* found in New Zealand, and on Chatham, Bounty, Antipodes, Snares, Auckland and Campbell islands; *rogersi* found in Australia, New Guinea and Indonesia; and *pelewensis* found in New Guinea, Vanuatu, New Caledonia, Fiji, Solomon Islands and French Polynesia. The subspecies *pelewensis* is markedly smaller (Amadon, 1943a), but *superciliosa* and *rogersi* cannot be differentiated morphologically. However, recent DNA analyses have found 2 divergent lineages in these latter two taxa: one found only in New Zealand, the other found throughout Australia and New Zealand (Rhymer et al., 2004). Pacific black ducks on Lord Howe Island (159°05‘ E, 31°33‘ S) occur at the geographic intersection of all 3 subspecies (Figure 5.1) and their lineage is unknown. Pacific black duck have been observed on Lord Howe Island since 1852 (MacDonald, 1853). Breeding is likely although records are infrequent: 1887 (Australian Museum records cited in McAllan et al., 2004), 1941-1945 (Hindwood and Cunningham, 1950) and 1971 (Rogers, 1972). A flock of 100 Pacific black ducks were observed in 1956, but this population subsequently declined (McKean and Hindwood, 1965).

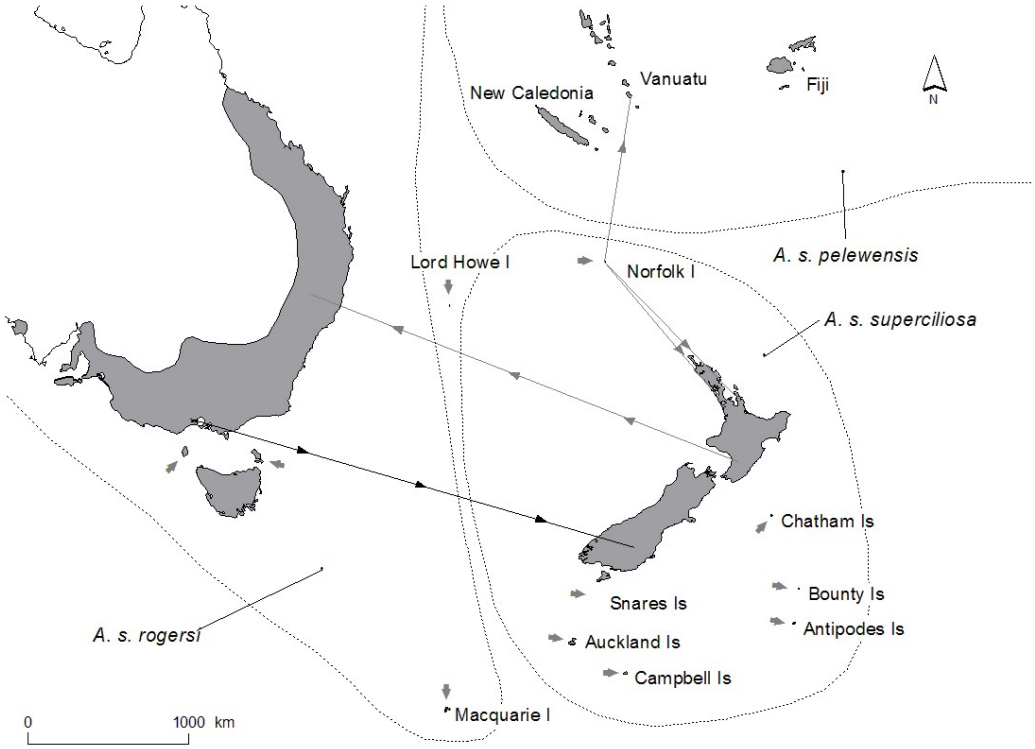

Figure 5.1. Movements and distribution of Mallard (grey arrows, grey shading) and Pacific Black Duck (black arrows, black dotted lines) in the South Pacific. Source: Norman 1973; Hermes et al., 1986; Marchant & Higgins 1990; Barrett et al., 2003; Robertson et al., 2007; ABBBS 2008. Distribution of *A. superciliosa* subspecies was taken from Rhymer et al., (2004). Note however that Amadon (1943a) and Frith (1982) included the birds of Macquarie Island within the distribution of *A. s. superciliosa*.

Mallard (*Anas platyrhynchos*) are native to the Holarctic and were introduced to Australia and New Zealand in the late 1860's (Lever, 1987). Hybridisation between Pacific black duck and mallard is common throughout New Zealand (Gillespie, 1985) to the extent that the genetic integrity of the Pacific black duck has been compromised (Rhymer et al., 1994). By 1982 only 4.5% pure Pacific black duck remained in Otago (Gillespie, 1985), and they now are thought to persist only in isolated, non-urban areas in New Zealand. Rhymer et al., (2004) suggested mallard should be eradicated from the Chatham Islands, and pure Pacific black duck be introduced to preserve its genetic integrity. Although rarely

quantified outside New Zealand, hybridisation between these two species also occurs elsewhere in the South Pacific, with the mallard implicated as a threat to the Pacific black duck where they co-occur (cf. Braithwaite and Miller, 1975; Williams and Basse, 2006),. Mallard were first recorded on Lord Howe Island in 1963 (McKean and Hindwood, 1965), and soon began hybridising with Pacific black duck (Rogers, 1976). The initial sightings of mallard coincide with major attempts to introduce this species to New Zealand: by 1963 over 20,000 mallard were released throughout the North and South Islands (McDowall, 1994). Mallard appear more sedentary than other ducks, although banding records demonstrate some long-range dispersal (Figure 5.1), and it is possible mallards on Lord Howe originated from New Zealand. Since 1975 reports of Pacific black duck on Lord Howe Island are likely to have been hybrids with mallards (Hutton, 1991). While Pacific black duck and mallard have been observed on Lord Howe Island since 1963 (McKean and Hindwood, 1965), there is no current information on their distribution and abundance, and few attempts have been made to manage these populations.

This paper quantifies the extent of hybridisation between Pacific black duck and mallard on Lord Howe Island, provides information on their abundance, current distribution, habitat use, and investigates the potential for their management.

5.2 Methods

Twenty-two systematic surveys of ducks were conducted in October 2007. All ducks were recorded along a standardised route from Ned's Beach and Old Settlement through to Evies and Kings Beach (Figure 5.3). Flock size, habitat type, location, activity, sex and age classes of birds were recorded. Detailed observations of plumage on 86 ducks were used to quantify the degree of hybridisation. This occurred both in the field (n=32 ducks) and with captured and shot individuals (n=54 ducks). Gillespie's (1985) seven point scoring system was used to differentiate phenotypic characteristics of the two species. Individuals with a score of 0-9 were considered Pacific black duck; scores of 10-24 were considered hybrids; and scores of 25-35 were considered mallard. The hybrid score was also separated into Pacific black duck-like hybrids (10-14), intermediate hybrids (15-19) and mallard-like hybrids (20-24). The minimum number of ducks known to be alive was also calculated by differentiating, when possible, individuals and groups during repeated counts using differences in age and plumage.

To trial methods for future management of ducks on Lord Howe Island we removed mallards and hybrids by trapping, shooting and hand capture according to ethics approval (Animal Research Authority ORA 05/019) and standard operating procedures (Sharp and Saunders, 2004b, a, c). Surveys of duck abundance and distribution were conducted before (n=10), during (n=5) and after (n=7) control measures were implemented. The timing and frequency of operation, and costs and labour was also recorded. Eight funnel entrance cage traps (approximately 1800 x 900 x 900 mm) and a pull net were used following approved procedures for pre-feeding, timing, trap operation, and handling, care and euthanasia of birds; and safety and legal considerations (Sharp and Saunders 2004a, b, c; Lukins and Tracey 2013). Baiting with bread and poultry layer mash occurred for 6 days prior to setting the traps. Trapped birds were removed and placed in a plastic holding box. Euthanasia was performed using cervical dislocation and inhalation of carbon dioxide (CO_2) following standard operating procedures (Sharp and Saunders, 2004a, b, c). Non-target species were released at the capture location. To avoid dispersing flocks, a .22 calibre rifle with silencer was used for shooting. Juveniles and chicks were also opportunistically captured by hand or handheld net. The percent reduction in mallards and mallard hybrids was estimated using 2 indices (standardised counts and minimum number alive) before and after management. An additional abundance estimate was calculated using index-manipulation-index (Riney, 1957; Caughley, 1980), with Eberhardt's (1982) variance estimate. Pearson's chi-squared tests were used to test habitat preferences separately for terrestrial and aquatic environments, using expected numbers estimated from the proportions of each habitat available within the searched area.

5.3 Results

Current extent of hybridisation

No birds were classified as pure Pacific black duck, despite sampling over 90% of the entire population (86 birds classified out of approximately 100 ducks present, see below). Pacific black ducks appear to have hybridised extensively with mallards, with only 2% of birds classified as Pacific black duck-like hybrids, 17% as intermediate hybrids, 41% as mallard-like hybrids and 40% as pure mallard. The mean phenotypic score was 24.2 (*se*= 0.59, range 1134, *n*=86), which is the upper limit of mallard-like hybrids (Figure 5.2).

Independent genetic study supports these phenotypic classifications, with sequencing of 5'-end of the mitochondrial DNA control region using feather samples revealing significantly higher hybridisation scores for Mallard compared with than Pacific Black Duck genotypes, and low genetic diversity (P=0.019, n= 44; Nucleotide diversity (π) 0.0004 ± 0.0005. Haplotype diversity (H) 0.279 ± 0.124; Lord Howe Island, Guay and Tracey, unpublished data; vs Nucleotide diversity (π) 0.0120 ± 0.0062, Haplotype diversity (H) 0.985 ± 0.005 of Mallards from the Western Palearctic, Kulikova et al. 2005)

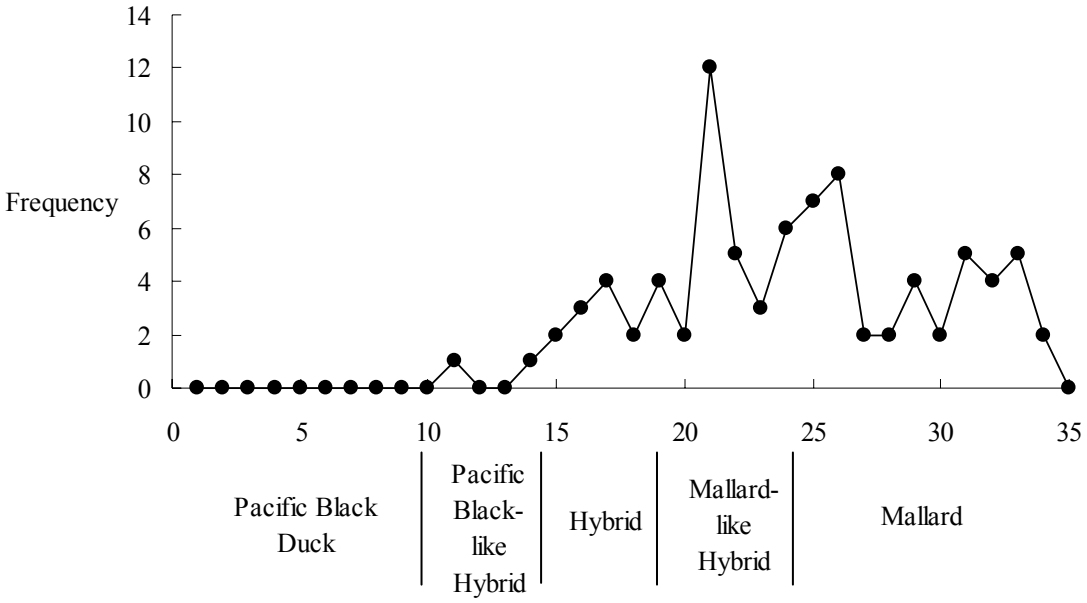

Figure 5.2. Frequency of phenotypic scores used to evaluate hybridisation between Mallard and Pacific Black Duck on Lord Howe Island.

Abundance, distribution and ecology of ducks on Lord Howe Island

Prior to management, the mean number of ducks observed per survey was 52.1 (se= 8.7, n=10 surveys) and the total number estimated using the index-manipulation-index method was 100.4 (se= 11.7). The minimum number known to be alive prior to culling was 98 ducks.

Ducks were most commonly observed on the golf course (31%, n=906 groups of ducks), Ned's Beach (25%) and Johnsons Creek area (17%; Figure 5.3). They also occurred in the

vicinity of Old Settlement (10%) and occasionally in residential areas and along roads (2%). Ducks regularly foraged on mown grass and pasture paddocks surrounding the airstrip (15%). None were observed in forested habitat, despite 1.58 km^2 (67%) of forest occurring within the search area. Following rainfall, temporary waterholes in pastures attracted groups of ducks. Similar behaviour was observed in fresh water pools formed in drainage lines. Larger family groups of unfledged ducks were resident near permanent dams within the golf course.

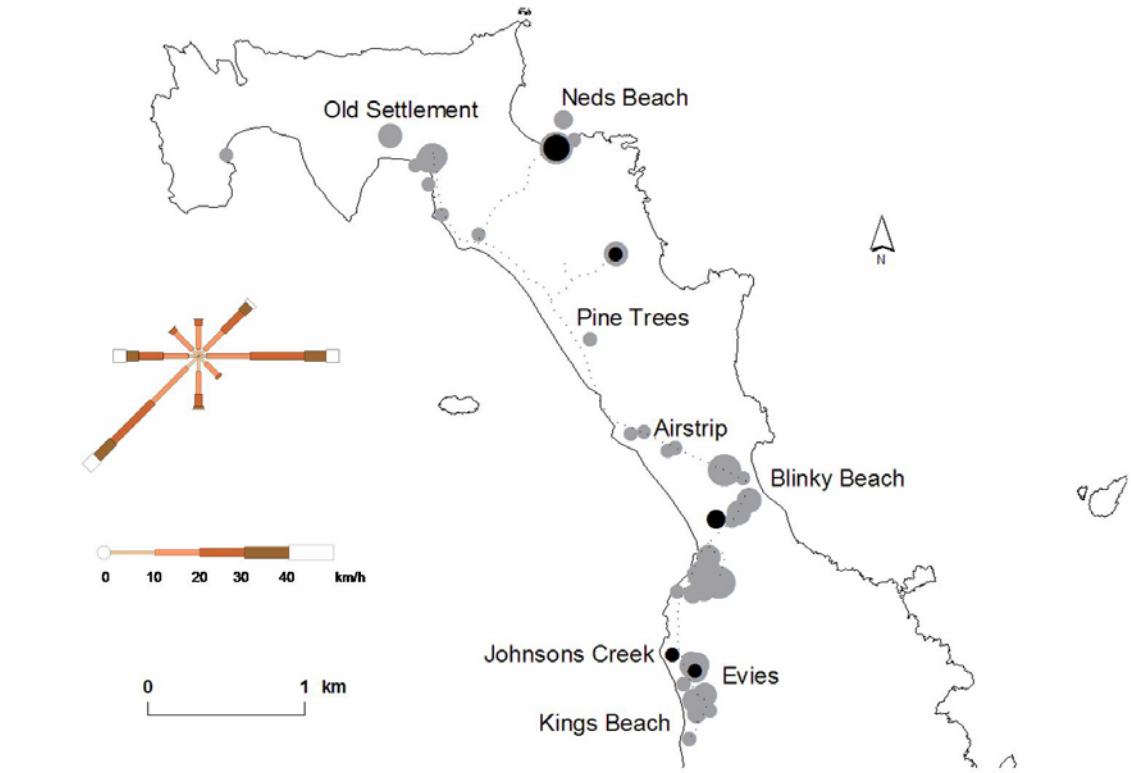

Figure 5.3. Distribution and abundance of Mallard x Pacific Black Duck hybrids on Lord Howe Island before (●) and after (●) control 8th- 18th October 2007. Flock size: 1-3, 4-6, 7-11, 12-18, 19-26 (increasing dot size). Observation route: (·····). The wind rose shows the long term average wind speed and direction from records taken at Lord Howe Island airport between 1989-2006 (n=3,933 observations Source: Australian Government Bureau of Meteorology).

When classified according to terrestrial habitat use, ducks preferred areas with mown grass (45%, n=654 groups of ducks, P<0.001, χ^2=3254.7) to grazed grass (25%) or the

revegetation area (11%). The beach was also used (18%), primarily at Ned's Beach where people feed fish. Observations of ducks on water indicated a strong preference for freshwater (72%, n=275 groups of ducks, P<0.001, χ^2=11,586), but ducks were also regularly observed in the ocean (17%) and estuaries (11%). Ducks spent the majority of their time standing (32%, n=692), swimming (21%) and walking (18%). Other activities included foraging (11%), resting (8%), flying (2%) and preening (1%). Breeding was observed in October, with 10 clutches observed (mean=7.7 eggs per clutch, *se*=0.78, range 412, n=10): nests were built close to fresh water in dense clumps of long grass. Recorded weights for mallard (male: mean=1040 grams, *se*=23.7, range 9101170, n=12; female: mean=930 grams, *se*=42.9, range 800-1040, n=6) were larger than those of the hybrids (female: mean=897 grams, *se*=19.6, range 730-990, n=13).

Control of mallards and hybrids

Seventy-two birds were captured and euthanased between 14-18 October 2007. The majority of these birds were shot (43%). Hand capture was the most cost-efficient technique (AUS$3.50/bird, cf. trapping AUS$19/bird and shooting AUS$22/bird) but was only used to capture juveniles and chicks when shooting adults. Trapping was hampered by the disturbance of traps by the public, and ducks did not become acclimatised to traps in some locations or bait was consumed by non-target species. Thirty-two birds of 5 non-target species were captured in traps, sampled for diseases and released at the capture location including banded rail (*Gallirallus philippensis*) (15), pukeko (*Porphyrio porphyrio*) (11), Lord Howe woodhen (*Gallirallus sylvestris*) (3), magpie lark (*Grallina cyanoleuca*) (2), and blackbird (*Turdus merula*) (1). Using the index-manipulation-index method the combined duck population before management was estimated to be 100.4 birds (*se*=11.69, n=10 surveys), with an estimate of only 28 birds after the cull (*se*=3.3). Management significantly reduced the distribution of ducks (Figure 5.3) and the minimum number alive after management was 26 birds. Using the mean number of ducks observed after management of 14.8 birds (*se*=6.18, n=4), the reduction was estimated as 71.7%. Using the minimum number of ducks known to be alive (98 before, 26 after), the reduction was estimated as 73.5%.

5.4 Discussion

Abundance – past and present

The decline in number of Pacific black duck on Lord Howe Island reported after 1956 may have been a result of hunting by humans and/or predation by cats, and was concomitant with the first records of mallard on the island in 1963 (McKean and Hindwood, 1965). Birds, including ducks, were a principal source of food for early island inhabitants, and hunting continued until at least the 1970s (Hutton, 1991). It is likely that Pacific black duck were hunted although it is unknown how many were killed. Feral cats (*Felis silvestris catus*) and possibly introduced pigs (*Sus scrofa*), may also have limited duck populations, although both were subsequently eradicated by 1981(Hutton, 1991). Pukeko, king fish (*Seriola lalandi*) and Lord Howe Island Currawong (*Strepera graculina crissalis*) are also known to prey upon ducklings (C. Haselden, personal observation; Hutton, 1991) and may also limit recruitment.

Hybridisation between Pacific black duck and mallard on Lord Howe Island was first reported in 1975 (Rogers, 1976). The total number of ducks has gradually increased from 50 to 60 birds reported in 1978 (Hutton, 1991) to the current prior-control population of about 100 birds, probably due to the provision of permanent water, creation of open lawn and grazing, and feeding by visitors and residents. Our survey revealed that all of these ducks are either mallards or hybrids. The extent of hybridisation on Lord Howe Island is thus even more advanced than in New Zealand (51% hybrids, 4.5% pure Pacific black duck; Gillespie, 1985). Mallard x Pacific black duck hybrids on Lord Howe Island were initially found to occupy areas below Mt Lidgbird and Blinky Beach swamp (Hutton, 1991). They were fed at Pine Trees Lodge and by 1987 began to regularly visit Ned's Beach where bread is fed to fish (Hutton, 1991). These hybrids are now common in developed parts of the island.

The origin of both Pacific black duck and mallard on Lord Howe Island is uncertain, but there are no records of deliberate introductions, and both species have the capacity to colonise new areas, either from the Australian mainland (Port Macquarie 586 km), New Zealand (1304 km), Norfolk Island (898 km), or New Caledonia (1258 km). Mallards are more likely to have arrived from New Zealand or Norfolk Island than the Australian mainland, as they are abundant in New Zealand (Gillespie, 1985) and banding has

confirmed their movements from Norfolk Island to New Zealand, Vanuatu and New Caledonia (Figure 5.1). However, the prevailing wind on Lord Howe Island is from the south–west during winter and spring (Figure 5.3), and it is possible either species could also travel from mainland Australia at this time of year.

Birds classified as mallard were heavier than hybrids, but unusually were lighter than both mallard and Pacific black duck in New Zealand (Balham, 1952), Australia (Miller, 1971; Braithwaite and Miller, 1975), and North America (Kortright, 1942). This may indicate that the original source of *A. superciliosa* was from the north (*pelewensis*) rather than west (*rogersi*) or south (*superciliosa*). The weights recorded for *A. superciliosa pelewensis* were considerably lighter (Amadon, 1943a) than the weights of mallard or hybrids in the current study. The impacts of mallards in a broader ecological context (e.g. Hone 2007) should also be considered. For example hybrids and mallards may occupy a similar ecological niche to Pacific black duck on the island. However, the most direct and immediate threat of mallard on Lord Howe Island is the decline and evident extinction of resident Pacific black duck. Evidence from this study suggests that the introduced mallard has eliminated the Pacific black duck previously present. Existing hybrids are also likely to continue to suppress any arriving Pacific black ducks. Williams and Basse (2006) suggest that mallard will soon permanently displace the Pacific black duck throughout New Zealand as a consequence of the mallard's greater survival, fecundity, physical domination, and willingness to exploit disturbed environments. Hybridisation with mallard has also been implicated as a major threat to anatids in other countries, including Canada and the United States (American black duck, *A. rubripes*, Ankney et al., 1989), Mexico (Mexican duck, *A. platyrhynchos diazi*; Hubbard, 1977; cf. Scott and Reynolds, 1984), Hawaii (Hawaiian duck, *A. wyvilliana*), and Madagascar (Meller's duck, *A. melleri,* Jones, 1996).

In the South Pacific co-occurrence of mallard and Pacific black duck has also been observed on Campbell Island (Bailey and Sorensen, 1962), Chatham Islands (Tennyson, 1998), Snares Islands (Miskelly et al., 2001), Auckland Island (Marchant and Higgins, 1990), Norfolk Island (Hermes et al., 1986) and Macquarie Island (Norman, 1990).

Management options

After our control program, a minimum of 21 adults (comprising 1 Pacific black duck-like hybrid, 3 intermediate hybrids, 9 mallard-like hybrids, and 8 mallard), 3 juveniles and 2

chicks were known to be alive. Bomford and O'Brien (1995) outline 6 criteria for successful eradication of a vertebrate pest. We believe that 5 of these criteria can be satisfied for mallard on Lord Howe Island: all animals are at risk by at least one method of control, rate of removal can exceed the rate of increase at all densities, the population can be monitored at all densities, discounted benefit-cost analysis is likely to favour eradication, and there is a suitable socio-political environment conducive to eradication. There is limited information on the benefits and costs of control, which are likely to increase as the population decreases. However, we believe the benefit-cost criteria are justified on the basis that control options are relatively inexpensive (currently AUS$3.50-$22/bird) compared with the potential benefits of the re-establishment of the native Pacific black duck. The remaining criterion, which stipulates that immigration is zero, is unlikely to be met for mallard or hybrids. However, on the basis of previous records of introductions (McKean and Hindwood, 1965; Rogers, 1976; Ray Shick, cited in Hutton, 1991), immigration is likely to be infrequent and preventing their re-establishment is considered manageable.

Although we consider the eradication of mallards and their hybrids on Lord Howe Island feasible, it would not be straightforward. In particular, disturbance of traps by the public and the tendency of birds to become flighty may make removal of the last few individuals problematic, labour intensive and therefore expensive. For these reasons, a hand delivered poisoning campaign following free-feeding may be a feasible option, and as an adjunct to targeted shooting, maybe effective in achieving eradication. Alphachloralose, a soporific, is considered the most humane avicide (Tracey et al., 2007), has been used for anatids in the United States (Woronecki et al., 1990; Woronecki et al., 1992) and is currently registered for use in Tasmania. However, an assessment of the non-target risks of poisoning and measures to limit these risks would need to be conducted before using this method. On welfare grounds, control should be implemented before breeding (Oct, this study; Jul-Nov, Hutton, 1991; Sep-Oct, McAllan et al., 2004). If breeding has commenced, hand capture of ducklings and juveniles before fledging, when they are less mobile, is a priority as subsequent control is more expensive and time consuming. Given the current distribution of mallard in the South Pacific and their capacity to travel large distances, re-colonisations following eradication are likely to occur. Equally the arrival of Pacific Black Ducks are likely and deliberate re-introductions of this species could be considered. If the eradication of mallards is pursued, ongoing monitoring and management will be necessary

to prevent their re-establishment and to protect the genetic integrity of any future population of Pacific black ducks.

PART C: HEALTH IMPACTS

Preface

Part C considers health impacts of birds using avian influenza in Australia as a case study. Since the first cases of H5N1 HPAI in humans in 1997 in Hong Kong (Xu et al., 1999), avian influenza has become internationally recognised by the public health practitioners, the animal health community and the general public. This virus is considered the precursor to subsequent major epizootics in 2001-2 (Guan et al., 2004) and 2003-4 (Li et al., 2004). By March 2004 epizootics were confirmed in China, Cambodia, Indonesia, Japan, Laos, South Korea, Taiwan, Thailand and Vietnam. Epizootics of H5N1 in poultry have since occurred throughout Asia (Figure 5.1), Europe and Africa, and its eradication is considered unlikely (Li et al., 2004; Sims and Narrod, 2009). H5N1 has also caused disease and death in humans (Claas et al., 1998; Subbarao et al., 1998; Yuen et al., 1998) via avian-to-human transmission.

The potential transmission of the H5N1, and other influenza A viruses from Asia to Australia via wild birds is of concern. There are many bird species known to undertake movements between Asia and Australia; the species involved, their movement behaviour, ecology and susceptibility to disease are all of importance when assessing the risks of avian influenza in Australia. In chapter 5 I review the epidemiology of avian influenza in wild birds, modes of transmission, and examine the factors influencing the prevalence of avian influenza in Australia's wild birds. In chapter 6 I undertake more detailed analysis of the abundance, movements and breeding ecology of Australia's Anseriformes in relation to the prevalence of low-pathogenicity avian influenza (LPAI) and provide risk profiles to improve the efficiency and relevance of wild-bird surveillance.

6. The role of wild birds in the transmission of avian influenza for Australia: an ecological perspective

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Abstract

Waterbirds, particularly Anatidae are natural reservoirs for low pathogenic avian influenza (LPAI) and have been implicated as the primary source of infection in outbreaks of highly pathogenic avian influenza (HPAI). An understanding of the movements of birds and the ecology of avian influenza viruses within the wild bird population is essential in assessing the risks to human health and production industries. Marked differences in the movements of Australian birds from those of the northern Hemisphere emphasises the danger of generalising trends of disease prevalence to Australian conditions. Populations of Anatidae in Australia are not migratory as they are in the northern Hemisphere, but rather display typical nomadic traits sometimes moving large distances across continental Australia in response to flooding or drought. There is little known regular interchange of anatids between Australia and Asia. In contrast, species such as shorebirds and some seabirds are annual migrants to Australia along recognised flyways from breeding grounds in the northern Hemisphere. Movement into Australia by these species mainly occurs from the north-west and along the east coast over the Pacific Ocean. These species primarily arrive during the Australian spring and form large aggregations along the coastline and on inland wetlands. Other Australian migratory species (passerines, bee-eaters, dollar-birds, cuckoos, doves) regularly move to and from Asia through the Torres Strait Islands. The disease

status of these birds is unknown. The movements of some species, particularly Anatidae and Ardeidae, which have ranges including Australia and regions where the virus is known to occur, have been poorly studied and there is potential for introduction of avian influenza subtypes via this route. Avian influenza viruses are highly unpredictable and have an ability to undergo re-assortment to more pathogenic forms. There is insufficient knowledge of the epidemiology and transmission of these viruses in Australia and broad-scale surveillance of wild birds is logistically difficult. Long-term studies of anatids that co-habit with Charadriiformes are recommended. This would provide an indication of the spatial and temporal patterns of subtypes entering Australia and improve our understanding of the ecology of endemic viruses. Until such time as these data become available, Australia's preparedness for avian influenza must focus on biosecurity at the wild bird-poultry interface.

6.1 Introduction

The biology and ecology of avian influenza viruses have previously been reviewed (Alexander, 1993). Influenza viruses are members of the Family Orthomyxoviridae and are characterised into types A, B or C on the basis of the antigenic character of the internal nucleoprotein antigen. Avian influenza is an infectious disease of birds caused by type A strains of the influenza virus (World Health Organisation Expert Committee, 1980). Only influenza A viruses have been isolated from avian species. The disease occurs worldwide and was first identified in Italy more than 100 years ago (Alexander, 1987). Avian influenza viruses normally do not infect species other than birds, but have been recorded infrequently in a range of other animal species including humans (Hinshaw et al., 1981; Alexander, 1982; Claas et al., 1998; Katz, 2003).

Influenza A viruses are divided into subtypes determined by haemagglutinin (H) and neuraminidase (N) antigens. At present, 15 H subtypes and 9 N subtypes have been identified. Each virus has one of each subtype in any combination (Animal Health Australia, 2003). The reservoir for all avian influenza virus haemagglutinin (H) and neuraminidase (N) subtypes is aquatic birds, particularly waterfowl (Suss et al., 1994), in which they multiply in the gastrointestinal tract producing large amounts of virus (Webster et al., 1978; Hinshaw et al., 1980) usually without producing clinical signs (Kida et al., 1980). In this environment, new combinations of H and N genes are generated and dispersed (Scholtissek et al., 1993). This process of exchanging genes between virus

strains is called re-assortment within influenza viruses and occurs when single cells of the host become co-infected with two genetically different viruses (Hinshaw et al., 1980). In wild waterbird hosts, the H and N subunits appear to be stable, and do not mutate (Sharp et al., 1997) like they do when the viruses infect domestic poultry and mammals. New virus combinations multiply readily in avian species and, in chickens and turkeys a proportion have a propensity to mutate and produce severe disease which in turn produce epizootics in poultry enterprises (Animal Health Australia, 2003).

Infection in birds causes a wide spectrum of symptoms, and viruses can be divided into two groups according to their pathogenicity (Office International Epizooties, 2001). Some forms of these viruses, known as highly pathogenic avian influenza (HPAI), can cause severe illness and mortality approaching 100% (Alexander, 1993; Swayne and Suarez, 2000). However, most strains of the virus are non-virulent, do not produce clinical signs or cause only mild respiratory or reproductive disease. These are known as low pathogenic avian influenza (LPAI) viruses which are commonly isolated from wild birds, particularly Anseriformes (swans, ducks and geese) (Slemons and Easterday, 1972; Stallknecht and Shane, 1988). Highly pathogenic influenza viruses, however, are not maintained by wild bird populations, but are occasionally isolated from wild birds during outbreaks in domestic poultry (Nestorowicz et al., 1987). The ability of LPAI to mutate into HPAI (Perdue et al., 1998), particularly in poultry, and the diversity of viruses circulating in wild bird populations (Webster et al., 1992) emphasises the potential importance of wild birds as a primary source of infection.

Epizootics of avian influenza may occur when a HPAI virus (with either a H5 or H7 haemagglutinin) is introduced to a naïve poultry population. Severe pandemics in humans occur when a major “antigenic shift” has occurred such as when the haemagglutinin is changed. Severe disease epidemics occur when there is “drift” with significant antigenic change in the haemagglutinin gene (Animal Health Australia, 2003). The presence of avian influenza viruses in wild birds thus has significance primarily for its potential to infect domestic poultry and humans, within which it can then undergo re-assortment to produce pathogenic forms (Webster et al., 1971, 1973). In addition, if humans are concurrently infected with both human and avian strains of influenza there is an increased risk of a new subtype emerging, which could result in the direct transmission between humans with the

possibility of a pandemic (Webster, 1998; Snacken et al., 1999; Baigent and McCauley, 2003; Katz, 2003).

There have been five known outbreaks of avian influenza in commercial bird flocks in Australia. Outbreaks occurred in 1976 (Turner, 1976), 1985 (Barr et al., 1986), and 1992 (Selleck et al., 1997) in Victoria; 1994 in Queensland (Westbury, 1998); and in 1997 in Tamworth New South Wales (Selleck et al., 2003). Viruses identified have all been of subtype H7 (H7N7, H7N3 and H7N4). The 2003-2004 Asian epidemic of HPAI (subtype H5N1) commenced in August 2003 and by March 2004 was confirmed in China, Cambodia, Indonesia, Japan, Laos, South Korea, Taiwan, Thailand and Vietnam. H5N1 has also caused disease and death in humans (Claas et al., 1998; Subbarao et al., 1998; Yuen et al., 1998) via direct avian-to-human transmission. Asia is of interest in particular, because of the frequency and distribution of highly pathogenic avian influenza (HPAI) epizootics (Figure 6.1) over an extended time period, and a range of other factors, such as high population density, poultry density and high levels of poultry- human interaction.

The potential transmission of the H5N1, and other influenza A viruses from Asia to other countries via wild birds is of concern. There are many bird species known to undertake movements between Asia and Australia; the species involved, their movement behaviour, ecology and susceptibility to disease are all of importance when assessing the risk of introducing foreign disease into Australia. The objective of this paper is to review the movements of wild birds between Asia and Australia, investigate their role in the transmission of avian influenza, and suggest ways in which Australia's management of avian influenza viruses associated with wild birds can be improved.

6.2 Movements of birds between Australia and Asia

Movements of wild birds into Australia from Asia occurs every year with the arrival of large flocks of migratory shorebirds and the movement of other species between the archipelagos of south-east Asia and northern Australia. Moreover, some species have distributions that extend to New Guinea and parts of south-east Asia were avian influenza is known to occur (Figure 6.1). Bird species known to travel between Asia and Australia are listed in Appendix II and discussed below.

Ducks, geese and magpie geese (Anseriformes)

Avian influenza has been isolated from species of ducks and geese (Anatidae) more than any other avian family, but is unknown in magpie geese (Anseranatidae). Members of the Anatidae family are ubiquitous throughout Australia, but unlike their northern Hemisphere counterparts (Hestbeck et al., 1991) most populations do not undertake predictable migrations associated with seasonal changes in resource availability (Kingsford and Norman, 2002 for review). Instead, movements of waterfowl in Australia are less predictable and many populations are nomadic—moving large distances in response to prevailing climatic conditions such as flooding or drought (Lawler and Briggs, 1991). This is particularly characteristic of species that occur in arid and semi-arid regions where resources are localised, ephemeral and affected by aseasonal stochastic processes (Halse and Jaensch, 1989; Lawler et al., 1993; Roshier et al., 2001a). The movement and distribution of most waterbirds across the Australian continent is therefore largely determined by the distribution of surface waters in the dryland river systems and the many ephemeral lakes and water-bodies of inland Australia (Briggs, 1992; Lawler et al., 1993; Kingsford, 1995; Roshier et al., 2001b; Roshier et al., 2002). Several species are wide-ranging and dispersive over most of the Australian continent, and are known to occur on the islands immediately to our north or have a geographic range that extends from northern Australia to parts of south-east Asia.

Movements of waterfowl from Australia to Asia have not been well studied but are thought to be irregular and to occur mainly from northern Australia. Banding records have confirmed the occurrence of these movements by Grey Teal (*Anas gracilis*) (Frith, 1982). Grey Teal are among the most dispersive of Australian waterfowl and are widespread on inland wetlands and sheltered estuarine and marine waters (Marchant and Higgins, 1990). Grey Teal are capable of moving thousands of kilometres over several weeks (Frith, 1959, 1963) and have been recorded moving from south-eastern Australia to Western Australia and New Zealand (Frith, 1957; Mills, 1976; Frith, 1982). In northern Australia, Grey Teal numbers peak in the dry season and are virtually absent in the wet season (December-April) (Marchant and Higgins, 1990). Grey Teal numbers in northern Australia fluctuate markedly with the greatest concentrations occurring following irruptions of southern populations (Frith, 1982). When Grey Teal occur in large numbers on the northern coastal wetlands they are usually in poor condition and are regarded as not good eating by the

traditional owners (Peter Kristofferson, personal communication), suggesting that their occurrence in northern Australia is in part driven by adverse conditions elsewhere. This species is known to be a dry season visitor to islands in the Torres Strait (Draffan et al., 1983) and may be a frequent traveller between New Guinea and northern Australia.

The range of other waterfowl which extend to the floodplains of New Guinea include the Magpie Goose (*Anseranas semipalmata*), Wandering Whistling-duck (*Dendrocygna arcuata*), Plumed Whistling-Duck (*Dendrocygna eytoni*), Rajah Shelduck (*Tadorna radjah*), Cotton Pygmy-goose (*Nettapus coromandelianus*), Green Pygmy-goose (*Nettapus pulchellas*), Pacific Black Duck (*Anas superciliosa*) and Hardhead (*Athya australis*) (Marchant and Higgins, 1990; Wetlands International, 2002). Most of these species are dispersive with little known about the nature and extent of their movements (Marchant and Higgins, 1990). Only the Cotton Pygmy-goose is regarded as sedentary, while the Plumed Whistling-duck is regarded as a partial migrant within Australia with few records from New Guinea where it is a vagrant (Marchant and Higgins, 1990). The range of the Wandering Whistling-duck includes northern Australia, New Guinea, and Pacific, Indonesian and Philippine Islands (Marchant and Higgins, 1990; Wetlands International, 2002). The East Indonesian and northern New Guinea populations are taxonomically distinct from those in Australia and southern New Guinea (Wetlands International, 2002; Dickinson, 2003), which suggests that the Australian subspecies does not extend north of New Guinea. Australian populations of Rajah Shelduck (*Tadorna radjah rufitergum*) are also taxonomically distinct from those that occur in New Guinea and the Moluccas Islands of Indonesia (Marchant and Higgins, 1990; Wetlands International, 2002; Dickinson, 2003). The Australian subspecies is known to occur in the Torres Strait throughout the year and may move infrequently between Australia and southern New Guinea (Draffan et al., 1983). The range of the Pacific Black Duck (*Anas superciliosa*) includes southern Sumatra, Java, Sulawesi, New Guinea, New Britain and islands of the southwest Pacific Ocean (Marchant and Higgins, 1990). There are three recognised subspecies, *superciliosa* (Australia, southern New Guinea, New Zealand), *rogersi* (Indonesian region) and *pelewensis* (northern New Guinea and islands of the southwest Pacific) (Marchant and Higgins, 1990; Dickinson, 2003). Pacific Black Duck are largely sedentary on permanent wetlands or in regions that are well watered, although they are dispersive from inland wetlands in summer. Birds banded at Griffith in southern New South Wales have been recovered as far afield as Tasmania, Queensland and New Zealand, but more than half of

all banded birds were recovered within 100 km of the point of release (Frith, 1959). Historically, Hardhead are known to occur in New Guinea and parts of Indonesia (Marchant and Higgins, 1990), but the most recent worldwide survey of waterbird populations does not recognise a population outside Australia, apart from that of the subspecies *Aythya australis extima* on Vanuatu and New Caledonia (Wetlands International, 2002; Dickinson, 2003).

Although the movements of most Australian waterbirds are largely unpredictable or poorly known, some seasonal movement patterns are known for northern distributed species. Magpie Geese for example spread out onto floodplains of northern-Australia during the wet season then retreat to remnant wetlands in the dry season (Morton et al., 1990b). Similar patterns are evident in Wandering Whistling-duck and Green Pygmy-Goose (Marchant and Higgins, 1990). Magpie Geese and Wandering Whistling-duck have been recorded moving across Torres Strait into New Guinea (Ashford, 1979; Draffan et al., 1983), and Green Pygmy-Goose are a dry season visitor to the Torres Strait (Marchant and Higgins, 1990).

Shorebirds (Charadriiformes)

Shorebirds (Family Scolopaciadae and Charadriidae) migrate annually between the northern and southern Hemispheres via known routes or flyways (Thomas, 1970; Tulp et al., 1994). There are around 3 million birds consisting of 35 species which regularly migrate from Australia each year (Wetlands International, 2002). Most depart from Australia in March to breeding areas, some as far as north-eastern Siberia and Alaska, although some juvenile birds will remain in Australia (Lane, 1987). Migrants return in September and spend late spring and summer in coastal and inland Australia (Tulp et al., 1994) (Figure 6.1; Appendix II). Larger species migrate further and over greater continuous distances, some travelling to China before stopping. Smaller species and juveniles which do migrate have 'stop-over' areas in Asia and south-east Asia, and travel shorter distances. Eight major flyway routes have been defined for waders based on biological and geopolitical considerations (Asia-Pacific Migratory Waterbird Conservation, 2001). The East Asian-Australasian flyway is of relevance to Australia and highlights the importance of major routes into Australia in the north-west and along the central-east coastline (Figure 6.1). Major 'stop-over' locations for these flights include sites throughout Asia including provinces where HPAI has been confirmed during the

2003/2004 outbreaks (Figure 6.1). Interchange between Australia and Asia has been confirmed through sightings of flagged birds and recoveries of banded individuals (McClure, 1998). Avian influenza is found occasionally in Charadriiformes (Table 6.1) with frequent occurrence of the virus in some species during particular seasons. Of particular relevance is that Charadriiformes can congregate in extremely large concentrations on coastal floodplains and mainland wetlands (Morton et al., 1993), where they regularly interact with Anatidae (Morton et al., 1990a).

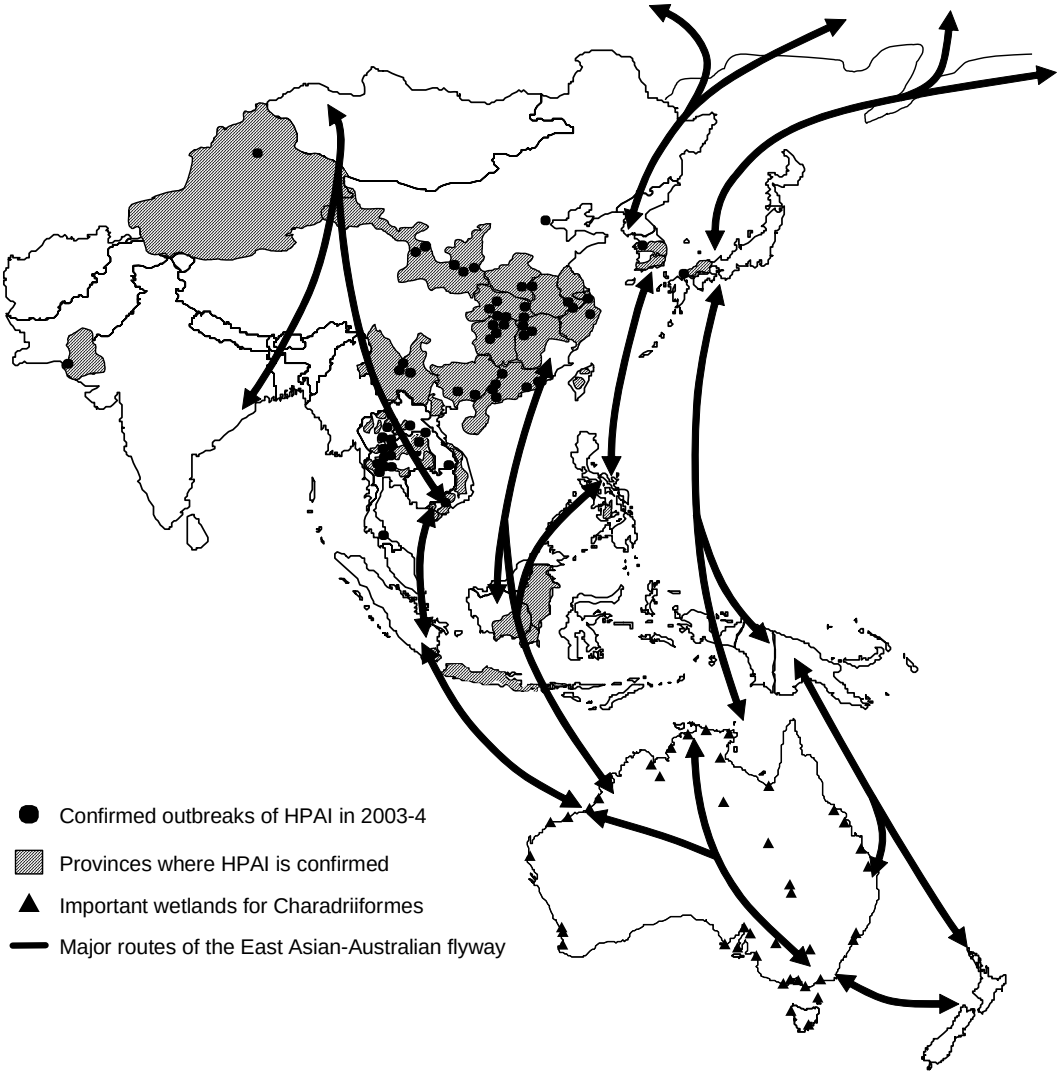

Figure 6.1. Major routes of the East Asian-Australian flyway in relation to the 2003-4 epizootic of highly pathogenic avian influenza (HPAI). Source: Wetlands International (Oceania) and Avian Influenza Map for Asia- situation on 18/02/04. (2004).

Grebes (Podicipediformes)

Grebes are distinct from any other groups of waterbirds (Sibley and Ahlquist, 1990), differing in morphology and behaviour as they are totally reliant on wetlands for food (Fjeldsa, 1988), protection (Hobbs, 1958) and nesting (Dann, 1981). Movements of grebes are poorly known. Most species are nomadic (Hobbs, 1956; Masters and Milhinch, 1974) with some migratory patterns suggested in northern populations (Marchant and Higgins, 1990). They are capable of travelling over large distances (Marchant and Higgins, 1990) but these movements are usually restricted to the Australian mainland. Movements into Asia are unknown but likely to be rare, despite evidence of breeding in the Torres Strait (Draffan et al., 1983). Australasian (*Tachybaptus novaehollandiae*) and Great-crested (*Podiceps cristatus*) Grebes are mainly solitary, although large flocks of several hundred can congregate in estuaries during winter (Wheeler, 1947). Hoary-headed Grebes (*Poliopcephalus poliocephallus*) are more gregarious with flocks up to several thousand (Fjeldsa, 1983), and up to 400 nests on a single wetland (Frith, 1976).

Albatrosses, petrels and shearwaters (Procellariiformes)

Procellariiformes spend most of their time on the open sea and only return to land to breed—mostly on off-shore islands. Their nesting sites in the Southern Ocean are among the remotest locations on Earth. Many species in this order are capable of extraordinarily long journeys across open water and some migrate to Australian waters in vast numbers. For example, the Short-tailed Shearwater (*Puffinus tenuirostris*) breed from October to February in south-eastern Australia and migrate in their millions to the north Pacific during March (Marshall and Serventy, 1961; Serventy, 1961). Many other shearwaters have similar movement patterns and breed in Australian waters in our summer (Harper and Kinsky, 1978). By contrast, Streaked Shearwater (*Calonectris leucomelas*) migrate from breeding islands off Japan and Korea and arrive in northern Australia in March (Gibson, 1975; Reilly, 1988; McClure, 1998).

Procellariiformes are generally restricted to the islands and open waters of the Pacific and Southern Oceans and are rarely observed closer than the continental shelf. Petrels in particular are seldom observed from the mainland and breed on Pacific islands considerable distances from the shoreline (Warham, 1990). The Shy Albatross (*Diomedea cauta*) is the only albatross to breed in Australia, also on off-shore islands (Harper and

Kinsky, 1978). Many species of this order regularly travel between Australia and Asia and are known to carry avian influenza (Downie and Laver, 1973).

Cormorants, darter and pelicans (terrestrial Pelecaniformes)

Cormorants, pelicans and the darter are nomadic and highly dispersive travelling large distances to exploit temporary resources wherever they occur (Llewellyn, 1983; Dorfman and Kingsford, 2001). Nesting of most species is colonial and frequently occurs with other species (Norman, 1974; Vestjens, 1977) increasing the likelihood of transmission of avian influenza between individuals and across species. Pelicans (*Pelecanus conspicillatus*) have been recorded moving into Papua New Guinea from South Australia (Marchant and Higgins, 1990) with large numbers recorded arriving in the Torres Strait (Draffan et al., 1983). Members of this order have worldwide distributions. The Darter (*Anhinga melanogaster*) has populations of four subspecies distributed across Australia, New Guinea, south-east Asia, India and Africa (Dickinson, 2003), including the Australian subspecies *Anhinga melanogaster novaehollandiae*. This subspecies is confined to Australia and New Guinea and is recognised by some authors as a distinct species (Wetlands International, 2002). Movements of Darter are poorly known but they are dispersive and banding studies have recorded movements of over 2000 km within Australia (Marchant and Higgins, 1990). Great Cormorants (*Phalacrocorax carbo*) have a worldwide distribution with the Australian subspecies *Phalacrocorax carbo novaehollandiae* confined to Australia and New Zealand (Dickinson, 2003). Some authors recognise two subspecies for Australia and New Zealand (Wetlands International, 2002). All four common cormorant species in Australia, Great Cormorant, Pied Cormorant (*Phalacrocorax varius*), Little Pied Cormorant (*Phalacrocorax melanoleucos*) and Little Black Cormorant (*Phalacrocorax sulcirostris*) are dispersive with some individuals (*P. carbo*) recorded moving beyond the Australian mainland following large breeding events inland (Marchant and Higgins, 1990). Flocks of Great and Little Black Cormorants have reached the islands of the Torres Strait and New Guinea (Geering et al., 1998), but movements off southern Australia are more common. The Little Pied and Little Black Cormorants are sympatric across much of their ranges in Australia, eastern Indonesia, south-west Pacific, New Guinea and New Zealand, while the Pied Cormorant is confined to Australia and New Zealand (Wetlands International, 2002). The Little Pied Cormorant has three subspecies and the Australian subspecies extends to Indonesia and Melanesia. Populations of the Little Black Cormorant are now considered to be a single species whose

range extends from southern Australia and New Zealand into New Guinea and eastern Indonesia (Dickinson, 2003). The nature and extent of cormorant movements between Australia and the northern parts of their range is largely unknown but likely to be infrequent.

Gannets, boobies, tropicbirds and frigatebirds (pelagic Pelecaniformes)

As partial migrants, some species of this order regularly pass through the oceans of Asia and South-Asia, especially during autumn (Marchant and Higgins, 1990). Similar to Procellariiformes, they are marine species and breed on off-shore islands (Nelson, 1978). Mainland sightings have been associated with summer cyclones blowing individuals inland (Morris, 1979). Boobies occur most commonly in tropical and sub-tropical waters and gannets prefer southern–temperate waters (Nelson, 1978; Brooke, 2004).

Egrets, heron, night heron, bitterns, stork, ibis, spoonbill (Ciconiiformes)

Egrets, Little Bittern (*Ixobrychus minutus*), ibis, White-faced Heron (*Egretta novaehollandiae*) and Royal Spoonbills (*Platalea regia*) are nomadic but also considered occasional dry winter (June-August) migrants to New Guinea (Hancock and Elliott, 1978; Finch, 1982; Draffan et al., 1983). Little is known of the nature and extent of movements of species of this order. Great Egret (*Ardea alba*), Rufous Night-heron (*Nycticorax caledonicus*) and White-faced Heron are known to be dispersive (Marchant and Higgins, 1990). Banding records have confirmed some interchange of Little Egret (*Egretta garzetta*) and Great Egret between Australia and New Guinea (Blakers et al., 1984; Marchant and Higgins, 1990). Striated Herons (*Butorides striatus*) are generally sedentary, although some Asian populations undertake a regular migration, for example to Christmas Island (Stokes et al., 1987). This species is highly differentiated taxonomically into 29 distinct subspecies or populations and there is little apparent movement between populations (Marchant and Higgins, 1990; Wetlands International, 2002). Large influxes of Rufous Night-heron in wetlands can occur after flooding (Hanscombe, 1915), with infrequent and erratic movements into New Guinea (Anonymous, 1977; Schodde and Mason, 1980; Draffan et al., 1983). Flocks of 50-100 Straw-necked Ibis (*Threskiornis spinicollis*) and 20-500 Australian White Ibis (*Threskiornis molucca*) have been observed in the Torres Strait (Draffan et al., 1983). Australian White Ibis nestlings banded in Australia and recovered in New Guinea confirm occasional movements further north (Carrick, 1962). The Glossy Ibis (*Plegadis falcinellus*) has a wide distribution that includes Australia, south-east Asia,

Africa and the east coast of North America (Wetlands International, 2002; Dickinson, 2003). Glossy Ibis are thought to be partial migrants in eastern Australia but movements elsewhere are thought to be erratic in response to rainfall (Marchant and Higgins, 1990). Black-necked Storks (*Ephippiorhynchus asiaticus*) are sedentary and have not been recorded moving north of the Australian mainland, except the occasional record in the Torres Strait (Draffan et al., 1983).

Other non-passerines

There are a variety of other non-passerines which regularly move between Asia and Australia. These include Rainbow Bee-eaters (*Merops ornatus*), Dollar-birds (*Eurystomus orientalis*), Brush (*Cuculus variolosus*) and Channel-billed (*Scythrops novaehollandiae*) Cuckoos, Common Koels (*Eudynamis scolopacea*), Superb Fruit Doves (*Ptilinopus superbus*), Pied Imperial Pigeons (*Ducula bicolor*) and the Kingfishers; Sacred (*Todiramphus sanctus*), Forest (*Halcyon macleayii*) and Buff-breasted Paradise (*Tanysiptera sylvia*). Most of these conduct regular migrations into Asia for the winter months (March-August).

The main flyway route for these species is via the Torres Strait islands and New Guinea (Higgins, 1999; Griffioen and Clarke, 2002). Some species remain in New Guinea through-out winter (April-August) (Schodde et al., 1975) while others continue further north. For example, Rainbow Bee-eaters travel through to Micronesia and Japan (Blakers et al., 1984); Common Koels to Indonesia and as far north as the Philippines (Rand and Gillard, 1967; Blakers et al., 1984); Channel-billed Cuckoos to southern Indonesia, the Bismarck Archipelago and New Guinea (Hindwood, 1953; Mason, 1981; Draffan et al., 1983); and Sacred Kingfishers to Timor, New Guinea, the Solomon islands and parts of Indonesia (Rand and Gillard, 1967; Bell, 1981; McClure, 1998). Some species display only partial migration, with individuals remaining in Australia throughout the year. This can vary with latitude, where southern populations of some species (e.g. Brush Cuckoos, Forest Kingfishers) are more migratory. All these species are observed migrating in flocks that are two orders of magnitude smaller than those of Charadriiformes (Lord, 1956; Hobbs, 1961; Warham, 1962; Gill, 1970; Lavery and Grimes, 1974; Draffan et al., 1983).

Passerines

Few passerines are known to move between Asia and Australia. Those species which have been recorded moving between the continents include Metallic Starlings (*Aplonis metallica*), Cicada Birds (*Coracina tenuirostris*), Spangled Drongos (*Dicrurus bracteatus*), Olive-backed Orioles (*Oriolus sagittatus*), and Brown-backed Honeyeaters (*Ramsayornis modestus*). Metallic Starlings, Cicada Birds and Spangled Drongos regularly migrate to south-east Asia during the winter months through islands of the Torres Strait (Barnard, 1911; Campbell and Barnard, 1917; Griffioen and Clarke, 2002). Southern populations of the Cicada Bird migrate north to New Guinea in autumn (Bell, 1982a; Draffan et al., 1983), while northern populations are partial migrants. Spangled Drongos exhibit more varied movements with some individuals conducting similar northward movements, while others move south during winter (Mayr and Rand, 1937; Bell, 1982b). Metallic Starlings, which are restricted to northern Queensland, roost and nest colonially in large numbers, with flocks of up to 5000 observed prior to migration (Blakers et al., 1984). Migrating Spangled Drongos usually form flocks of approximately 20 (Draffan et al., 1983), but larger flocks have been observed over Thursday Island (Blakers et al., 1984). Movements of Olive-backed Orioles and Brown-backed Honeyeaters into Asia are uncommon and irregular, apparently fluctuating with the availability of ripe fruit or nectar (Officer, 1964; Gill, 1970; Storr, 1973; Draffan et al., 1983).

6.3 Occurrence of avian influenza in wild birds

There are a large number and variety of influenza viruses maintained in wild bird populations. Avian influenza viruses have been isolated from more than 88 species of wild birds from 12 orders comprising most of the major families Stallknecht and Shane 1988 (Stallknecht and Shane, 1988; Alexander, 2000; Olsen et al., 2006; Tracey 2010) for review). The first isolation from wild birds occurred in South Africa from Common Terns (*Sterna hirundo*) in 1961 (Becker, 1966). An increase in surveillance during the late 1970s revealed ducks and geese (Anseriformes) as the main reservoir of the viruses, where prevalence exceeded 60% in some studies (Hinshaw et al., 1980). Overall prevalence rates estimated from 122,830 samples indicate around 7.9% of Anseriformes are infected with the virus at any one time (Table 6.1). However, many of these studies are only based on regular samples of Anseriformes and may be unrepresentative of region and species. In other studies Charadriiformes (shorebirds, plovers and lapwings) (Kawaoka et al., 1988) and spoonbills (Astorga et al., 1994) have also been found to have a high prevalence of the

virus, with isolation rates of up to 20% and 32% respectively. In Australia, prevalence of the virus is found to be much lower (Mackenzie et al., 1984; Mackenzie et al., 1985; Peroulis and O'Riley, 2004, Tracey et al., 2010). Isolation rates and subtypes vary considerably over time, region and between species (Kawaoka et al., 1988; Sharp et al., 1993). This has been identified for Charadriiformes where sampling along the Atlantic coast and the Gulf of Mexico revealed 78% of isolates from Ruddy Turnstones (*Arenaria interpres*), with concentrations of the virus during one season (spring) and in one location (Delaware Bay) (Hanson, 2003). Anatini tribes of Anseriformes also exhibit higher prevalence of avian influenza than other species of the same order (Stallknecht and Shane, 1988). Other species normally not associated with the maintenance of avian influenza viruses are also occasionally infected (Table 6.1; Stallknecht and Shane, 1988). This has also occurred during outbreaks of HPAI, for example, Starlings (*Sturnus vulgaris*) (Nestorowicz et al., 1987), ratites (Selleck et al., 2003) and flamingos, falcons and crows during the 2003-4 Asian epidemic.

Seasonal infection patterns have emerged in Anseriformes, with the greatest prevalence during late autumn and winter (Sinnecker et al., 1982; Halvorson et al., 1985). This trend is consistent with the timing of outbreaks of human influenza, but differs from the spring epidemics evident in Charadriiformes (Hanson, 2003). Movements and age of birds also appear to be important and correlated with seasonal effects. For example, a significantly higher prevalence of the virus was recorded for juvenile mallards before migrating south for the winter (Deibel et al., 1985; Hinshaw et al., 1985; Hinshaw et al., 1986).

Most subtypes have been detected in Australian wild birds (H1, H3, H4, H5, H6, H7, H11, H12) (Downie and Laver, 1973; Downie et al., 1977; Mackenzie et al., 1984; Mackenzie et al., 1985; Nestorowicz et al., 1987; Peroulis and O'Riley, 2004). In Australia, no quantitative links have been made to wild birds during the five previous HPAI outbreaks in poultry. In addition Australian isolates of HPAI have been found to be distinct from those in other parts of the world including Asia (Banks and Alexander 1997; see also Rohm et al., 1995 for an international perspective), which suggests an endemic rather than exotic source of infection. However, there is circumstantial evidence that waterfowl may have been involved in previous outbreaks; and sampling has been limited and on one occasion may have occurred too long after the epidemic (Westbury, 1998; Selleck et al., 2003). Direct and in-direct contact with waterfowl birds has been reported and has been suggested

as a potential cause of initial infection (Westbury, 1998). During the 1997 outbreak in Tamworth New South Wales, HPAI was isolated from an adjacent Emu (*Dromaius novaehollandiae*) farm which was suggested to have played a role in the transmission of the virus (Selleck et al., 2003). Some evidence also suggests the transmission of HPAI can occur between domestic poultry and passerine birds, following reports of the virus infection in starlings (Nestorowicz et al., 1987; Westbury, 1998).

The unpredictability inherent in avian influenza viruses; the variation in prevalence between species (e.g. Becker, 1967) and temporal and spatial variation in virus occurrence makes generalisations across families difficult. However, to allow targeted surveillance, four prevalence classes have been identified to describe the relative occurrence of avian influenza virus (Table 6.1). Information contained in Table 6.1 was derived from information gathered from within Australia (Downie and Laver, 1973; Downie et al., 1977; Mackenzie et al., 1984; Mackenzie et al., 1985; Nestorowicz et al., 1987; Peroulis and O'Riley, 2004; Tracey 2010) and review publications of studies conducted in the northern Hemisphere (Stallknecht and Shane 1988; Stallknecht 1998; Hanson 2003; Olsen et al., 2006). Of the 27 families known to move between Australia and Asia (Appendix II), avian influenza infection is suggested to commonly occur in Anatidae and Ardeidae (herons, egrets, night-herons and bitterns) and is occasional in Charadriidae (plovers, dotterels and lapwings), Laridae (skuas, jaegers, gulls and terns) and Scolopacidae (snipe, godwits, curlews, sandpipers, stints and phalaropes) (Table 6.1). The virus is rarely found in members of Threskiornithidae (ibis and spoonbills), Procellariidae (petrels, shearwaters and prions), Phalacrocoracidae (cormorants), Columbidae (pigeons and doves) and Sturnidae (starlings and mynas) families, and is unknown in the other 15 families (Table 6.1).

Table 6.1. The relative occurrence of avian influenza in families of birds known to move between Australia and Asia

Families were classified according to the relative prevalence of avian influenza (n=122,830 samples from wild birds) using a increasing scale: Unknown (0%), Rare (<1%), Occasional (1-5%), Common (>5%), derived from information in Downie and Laver, 1973 (1); Downie *et al*.1977 (2); Hanson 2003 (3); Kawaoka *et al.*1988 (4); Lipkind *et al.,* 1982 (5); Mackenzie *et al.,* 1984; 1985 (6); Morgan and Kelly 1990 (7); Olsen et al. 2006 (8); Peroulis and O'Riley 2004 (9); Romvary and Tanyi 1975 (10); Roslaya *et al.,* 1974 (11); Slemons *et al.,* 1973 (12); Stallknecht and Shane 1988 (13); and Tracey 2010 (14).

Order	Family	Common Family	Relative occurrence of avian influenza	Source
Anseriformes	Anseranatidae	Magpie Geese	Unknown	6
	Anatidae	Waterfowl	Common	6, 8, 9, 13,, 14
Procellariiformes	Procellariidae	Shearwaters/Petrels	Rare	1, 2, 6, 8, 14
	Hydrobatidae	Storm Petrels	Unknown	8, 14
Pelecaniformes	Phaethontidae	Tropicbirds	Unknown	
	Sulidae	Boobies/Gannets	Unknown	
	Phalacrocoracidae	Cormorants	Rare	2, 8, 14
	Pelecanidae	Pelicans	Unknown	8, 14
	Fregatidae	Frigatebirds	Unknown	
Ciconiiformes	Ardeidae	Herons/Bitterns	Common	2, 8, 11, 14
	Threskiornithidae	Ibises	Rare	2, 8, 14
Gruiformes	Rallidae	Rails	Unknown	
Charadriiformes	Scolopacidae	Turnstones/Sandpipers/Phalaropes	Occasional	3, 4, 8, 14
	Charadriidae	Plovers	Occasional	3, 4, 8, 14
	Glareolidae	Pratincole	Unknown	
	Laridae	Gulls/Terns	Occasional	3, 4, 8, 14
Columbiformes	Columbidae	Pigeons/Doves	Rare	10, 8, 14
Cuculiformes	Cuculidae	Cuckoos	Unknown	
Coraciiformes	Halcyonidae	Kingfishers	Unknown	
	Meropidae	Bee-eaters	Unknown	12
	Coraciidae	Dollarbird	Unknown	
Passeriformes	Meliphagidae	Honeyeaters	Unknown	
	Dicruridae	Drongoes	Unknown	
	Campephagidae	Cuckoo Shrikes	Unknown	
	Oriolidae	Orioles	Unknown	
	Sturnidae	Starlings	Rare	5, 7, 8, 14

6.4 Transmission

Examining the spread of avian influenza is difficult, hence most information on transmissibility is based on laboratory experiments (Alexander, 1993). Factors contributing to virus transmission are complex, and variability exists between subtypes, bird species and environmental factors.

Subtypes of avian (type A) influenza are identified by the combination of H (haemagglutinin) and N (neuraminidase) proteins (Office International Epizooties, 2001) and are important when considering the potential for transmission and mutation. All 15 subtypes are known to infect birds but in nearly all cases, only subtypes with H5 and H7 are known to mutate to the highly pathogenic form (cf. Laudert et al., 1993; Office International Epizooties 2001). HPAIs have been documented to arise from LPAI viruses (Perdue et al., 1998), but are not normally known to change subtypes during an outbreak. However, evidence suggests recombination can occur when birds are infected with multiple subtypes (Webster et al., 1973; Sharp et al., 1997; Hoffmann et al., 2000). Further, some evidence indicates viruses which are better adapted to avian populations have a demonstrated ability to prevent infections of other strains (Sharp et al., 1997). This may imply that wild birds which currently maintain well-adapted LPAI viruses are less likely to transmit HPAI.

In most cases mutation into highly virulent viruses takes place only in domestic poultry, which occurs after their exposure to LPAI viruses. The only known outbreak of HPAI in wild birds occurred in Common Terns in 1961 (Becker, 1966). Wild birds are implicated as important in this initial stage of transmission, but are not considered reservoirs of highly pathogenic strains. In comparison to wild birds, which do not normally show symptoms of disease, poultry are highly susceptible to H5 and H7 subtypes. Hence subtypes which do not appear to affect wild birds have caused fatal diseases in poultry and other domestic birds. Birds which become exposed and survive infection may excrete virus for up to 14 days, orally and in faeces (Kida et al., 1980). While this, in theory, may provide opportunity for wild birds to spread the virus over a large distance, there is no documented evidence of this occurring for HPAI viruses. In most cases secondary spread of HPAI has been associated with human activity, including live poultry markets (Panigrahy et al., 2002), rather than wild bird hosts (Wells, 1963; Alexander, 1993; Westbury, 1998; Swayne and Suarez, 2000).

Water is a likely medium for the transfer of non-virulent avian influenza and partially explains the high prevalence of the virus in Anseriformes and Charadriiformes which congregate in large numbers in wetlands. The virus can remain infective in freshwater lakes for 4 days at 22°C, over 30 days at 0°C (Webster et al., 1978), or up to 200 days at

17°C at higher concentrations (Stallknecht et al., 1990b). This indicates a potential role in the transfer of the virus to poultry via contaminated water supplies sourced from dams, wetlands and other waterbird refuges. The duration of infectivity of water is also shown to decrease with increased salinity and pH (Stallknecht et al., 1990a), which may have implications for the maintenance of the virus in shorebirds and seabirds, and the management of water used in poultry production.

In some studies, a high prevalence of the virus and antibodies have been recovered from the eggs of waterbirds (Narayan et al., 1969; Romvary et al., 1980); Cappucci et al., 1985). Breeding areas of Charadiiformes often involve large numbers of eggs at specific sites (Lane, 1987; Pringle, 1987), which may provide an opportunity for sampling for avian influenza. However, the role of eggs in the transmission or maintenance of the virus is unknown.

6.5 Avian influenza in Australia

The role of wild animals in the introduction, maintenance and transmission of disease is largely dependent on a range of ecological factors, including the distribution and density of susceptible wild animal disease hosts (Animal Health Australia, 2003). The risks associated with wild birds introducing H5N1 or other subtypes of avian influenza are virtually impossible to quantify with current information. There is insufficient knowledge of the epidemiology and transmission of avian influenza viruses and a lack of reliable information on the interchange of many birds between Asia and Australia, particularly of the Anatidae and Ardeidae. Moreover, avian influenza viruses are highly unpredictable and have a documented propensity for mutation. Review of current knowledge of bird movements and avian influenza in Australia is important for identifying the focus for future research and targeting species, timing and regions for surveillance.

Ninety-nine bird species are known to move between Asia and Australia (Appendix II). Sixty-three of these undertake frequent migration, 20 travel occasionally and 16 rarely visit. Shorebirds (Charadriiformes) regularly migrate to Asia, but are mainly aggregated along Australian coastlines and at specific inland wetlands. In contrast ducks and geese (Anseriformes) and other nomadic waterbirds are widely distributed but rarely move from the Australian mainland. Pelagic birds (Procellariiformes and Pelecaniformes) are annual or partial migrants, and are occasionally known to carry the virus but are rarely observed

inside the continental shelf. Other migratory species of northern and north-eastern Australia, travel through the Torres Strait Islands during winter but are unlikely to carry avian influenza. The risk to Australia appears to be in the association between shorebirds, which are potentially harbouring avian influenza viruses, and Australian ducks and geese. If infected, these ducks and geese could potentially spread virus to poultry farms as they disperse from coastal areas. If affected, poultry would then have the ability to transmit virus to humans. However, to date the transmission of avian influenza from poultry to humans is rare and has only been associated with a small number of viruses, mainly of Asian origin (Horimoto and Kawaoka, 2001; Baigent and McCauley, 2003; Katz, 2003).

By definition, highly pathogenic avian influenza (HPAI) has the potential for very serious and rapid spread, irrespective of borders, which is of serious socio-economic and public health consequence, and is of importance in the international trade of livestock and livestock products. In Australia, current procedures for the management of incursions of HPAI within the poultry industry involve eradication. The five previous outbreaks of HPAI in the Australian poultry industry were eradicated by 'stamping out' - a procedure involving the destruction of all potentially susceptible birds (Animal Health Australia, 2003). However, the destruction of wild birds is unlikely to be effective, useful or practical in preventing the spread of the virus. Management of the virus should instead focus on ensuring wild birds do not come into contact with domestic birds, either by direct contact or by contaminated water (Animal Health Australia, 2003). Prevention and control measures can minimise, or eliminate the risk from contaminated water (World Health Organisation 2004). For example, influenza viruses are relatively susceptible to disinfectants (chlorine) and heating, so boiling would also be effective (World Health Organisation 2007).

The Australian poultry industry is small in comparison to many other countries, including Hong Kong and China (Animal Quarantine Policy Branch 2001). The main areas within Australia for poultry production are usually sufficiently isolated from one another (Animal Health Australia, 1996) to provide some protection against widespread transmission of exotic disease. Where poultry (and susceptible animals) exhibit a contiguous or near-contiguous population, the risk of widespread disease transmission may increase substantially.

There are no surveillance or vaccination programs currently in place for endemic avian influenza in poultry. The chances of detecting avian influenza viruses in shorebirds appears small, however, in other studies, the chance of detection increases five-fold if waterbirds that are in contact with shorebirds are targeted for surveillance (Suss et al., 1994). A number of potential models could be used for surveillance of wild birds. Surveillance may be more effective if set up where waterbirds have a greater risk of interacting with poultry, such as around free-range poultry establishments, 'backyard' operations or where biosecurity measures are lacking. The interaction between these farms and other commercial operations is also important in understanding the persistence of avian influenza viruses and their contact with poultry. Surveillance in remote aggregations of waterbirds in Australia may therefore be less important than where concentrations of domestic poultry occur, for example, near capital cities and key regional areas of NSW, Victoria and Queensland (Animal Quarantine Policy Branch 2001).

As a result of their large population sizes, surveillance of wild birds is logistically difficult, and large sample sizes are required to provide statistically meaningful results. A more useful option might be to focus avian influenza work in wild birds on long-term, longitudinal studies in waterfowl, which share habitat with shorebirds (Suss et al., 1994). This would give an indication of the spatial and temporal patterns of subtypes entering Australia, and could act to significantly improve understanding of the ecology of these viruses within Australia.

6.6 Conclusions

An understanding of the ecology of the viruses within the wild bird population is essential in assessing the risks to human health and production industries. Long-term surveys for viruses in wild birds are required to improve our understanding of the prevalence of LPAI viruses and the role they play in the transmission of avian influenza to poultry and humans. These surveys should target Anatidae and Ardeidae where there are highest densities of Charadriiformes, and could be strategically coordinated between field naturalist societies, research groups (such as the Australian Wader Study Group) and government authorities. For detecting the introduction of exotic viruses, sampling should focus on coastal floodplains of the north-west, along the central-east coastline and other important wetlands for Charadriiformes (Figure 6.1). Knowledge of the interface between Anatidae and domestic poultry is needed for assessing the risks of virus transfer. The optimal time for

sampling would occur when shorebirds first arrive in Australia (August- October). This time period also coincides with suggestions that higher prevalence of avian influenza for these species occurs in spring. More accurate information is required on the movements of waterbirds between Asia and Australia, particularly Anatidiae and Ardeidae. This would aid our understanding of the importance of wild birds in introducing foreign subtypes of avian influenza as well as their potential to transmit other viruses, including Japanese encephalitis and Newcastle disease. Until such time as these data become available, Australia's preparedness against HPAI must focus on biosecurity at the wild bird-poultry interface.

7. Risk-based surveillance of avian influenza in Australia's wild birds

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Abstract

Context. The epidemiology of avian influenza and the ecology of wild birds are inextricably linked. An understanding of both is essential in assessing and managing the risks of highly pathogenic avian influenza (HPAI).

Aims. This project investigates the abundance, movements and breeding ecology of Australia's Anseriformes in relation to the prevalence of low-pathogenicity avian influenza (LPAI) and provides risk profiles to improve the efficiency and relevance of wild-bird surveillance.

Methods. Generalised linear models and analysis of variance were used to examine the determinants of Anseriformes abundance and movements in Australia, and the observed prevalence of LPAI in Australia (n = 33,139) and overseas (n = 93,344). Risk profiles were developed using poultry density, estimated LPAI prevalence, the abundance of Anseriformes, and the probability of Anseriformes moving from areas of HPAI epizootics.

Key results. Analysis of Australian wild-bird surveillance data strongly supports other studies that have found the prevalence of LPAI in wild birds to be much lower (1%) in Australia than that in other countries (4.7%). LPAI prevalence was highly variable among sampling periods and locations and significantly higher in dabbling ducks than in other

functional groups. Trends in Anseriformes movements, abundance and breeding are also variable, and correlated with rainfall, which could explain low prevalence and the failure to detect seasonal differences in LPAI in wild birds. Virus prevalence in faecal samples was significantly lower, whereas collecting faecal samples was 3–5 times less expensive and logistically simpler, than that of cloacal samples. Overall priority areas for on-going surveillance are provided for Australia.

Conclusions. Previous surveillance has occurred in high-priority areas, with the exception of Mareeba (North Queensland), Brisbane and Darwin, and has provided valuable information on the role of wild birds in maintaining avian influenza viruses. However, several practical considerations need to be addressed for future surveillance.

Implications. Long-term surveillance studies in wild birds in priority areas are required, which incorporate information on bird abundance, age, behaviour, breeding and movements, particularly for dabbling ducks. This is important to validate trends of LPAI prevalence, in understanding the main determinants for virus spread and persistence, and in predicting and managing future epizootics of HPAI in Australia.

7.1 Introduction

Highly pathogenic avian influenza (HPAI) has caused international concern during the past decade, particularly HPAI H5N1, because of its ability to infect and cause death in humans, its ability to cause significant mortality in wild birds (Liu et al., 2005), the longevity of HPAI circulation, the failure to isolate closely related precursor strains of low-pathogenicity avian influenza (LPAI) in wild birds (Mukhtar et al., 2007), and subsequent evolving viruses remaining highly pathogenic for poultry (Sims et al., 2005; Sims and Narrod, 2009).

Between 1997 and 2004, mutations in HPAI H5N1 were progressively becoming more lethal to birds and mammals and hardier in the environment (Chen et al., 2004). Although there is still debate as to whether an independent cycle of infection of HPAI H5N1 is present in wild birds (Feare, 2007; Wang et al., 2008), some evidence following wild-bird deaths suggests that wild birds, particularly anatids, can excrete virus without becoming ill

(Hulse-Post et al., 2005; Gaidet et al., 2008; Keawcharoen et al., 2008) and transmission appears to occur even if the virus is difficult to detect (Stallknecht and Brown, 2008).

HPAI H5N1 has now been circulating in close proximity to Australia for over 11 years and the likelihood of achieving eradication is considered low (Sims and Narrod, 2009). With the exception of Antarctica, Australia remains the only continent that has not had a reported occurrence of HPAI since 1997 (Sims and Turner, 2008). There has been several possible explanations for this (McCallum et al., 2008) including (1) enhanced biosecurity for the trade of live wild and domestic birds (2) low prevalence of LPAI H5 or H7 in Australia's wild birds (Haynes et al., 2009), (3) limited interchange of anatids between Australia and Asia (Tracey et al., 2004; McCallum et al., 2008), (4) few major waterbird breeding events and low waterbird abundance (Nebel et al., 2008), (5) low poultry-farm density (Westbury 1998; cf. Hamilton et al., 2009), and (6) high biosecurity of the commercial poultry industry.

An understanding of avian influenza epidemiology in wild birds is important in assessing and managing the risks of HPAI of any type. Many H and N subtypes of LPAI have been reported in Australia's wild birds, including H5 and H7 (Downie and Laver, 1973; Downie et al., 1977; Mackenzie et al., 1984; Mackenzie et al., 1985; Nestorowicz et al., 1987; Rohm et al., 1996; Peroulis and O'Riley, 2004; Hurt et al., 2006; Haynes et al., 2009). Anseriformes are the primary reservoir of LPAI (Stallknecht and Brown, 2008; Haynes et al., 2009), with high prevalence associated with foraging behaviour (Anas species), age, breeding and movements (Hinshaw et al., 1985; Olsen et al., 2006; Haynes et al., 2009); Munster and Fouchier 2009). Charadriiformes are also considered potentially important (Hurt et al., 2006) and regularly travel through infected areas (Tracey et al., 2004), although unique lineages of influenza viruses in Australia compared with viruses in Europe and the Americas (Banks and Alexander, 1997) suggest limited virus interchange via these species.

Although the importance of wild birds in avian influenza epidemiology is now widely accepted by the international animal health community (OIE, FAO), broad-scale surveillance is logistically difficult and costly because of the natural low prevalence. Initial surveillance in Australia followed a targeted approach to improve sampling efficiency (Tracey 2005; Warner et al., 2006). East et al., (2008a , 2008b) and Hamilton et al., (2009)

have also applied useful approaches to classify risks of avian influenza in Australia. Since initial surveillance, there have been significant advancements in avian influenza epidemiology, particularly for HPAI H5N1; improved information on the abundance and movements of Australian waterfowl, although many gaps in knowledge remain (McCallum et al., 2008); and a substantial increase in wild-bird surveillance for avian influenza in Australia and internationally. This information can be used to maximise the efficiency and relevance of avian influenza wild-bird surveillance in Australia. The present project investigates trends of avian influenza in wild birds and waterfowl abundance and movements, identifies high-risk areas for avian influenza in Australia and provides recommendations for surveillance.

7.2 Materials and methods

Analysis of Anseriformes movements and abundance

Generalised linear mixed models and analysis of variance were conducted using the *asreml* package (Butler et al., 2009) under R (R Core Team, 2013) to investigate the effects of the functional group, sex, age, the availability of permanent water (km^2 per 1/4° grid from Geoscience Australia and National Water Commission data on rivers, dams and inland water), season, seasonal rainfall classification (a Bureau of Meteorology classification, identifying summer-dominant, summer, uniform, winter, winter-dominant, or arid rainfall), and a range of rainfall co-variates (annual rainfall, mean annual rainfall, seasonal rainfall, rainfall in the previous 2 and 3 months, monthly rainfall) from the Bureau of Meteorology and interactions on the distance moved (log-transformed to remove heterogeneity of variance) by Anseriformes after banding. Site and sampling event were included as random variables to account for multiple records from the same locations and sampling periods. Stepwise regression was performed to remove non-significant terms ($P>0.05$). Wald tests were used to examine fixed effects and non-marginal terms not significant at the $P = 0.05$ level were dropped sequentially. Differences between terms for discrete variables are presented with 95% confidence intervals. A sampling event was defined as all captures occurring on the same site in the same month. Distances moved for Anseriformes were estimated as kilometres travelled from the capture location, using banding data from the Australian Bird and Bat Banding Scheme (www.environment.gov.au/biodiversity/science/abbbs/, accessed 1 January 2009). Anseriformes were aggregated into functional groups based on behaviour and feeding

habits (after Kingsford, 1991 and Roshier 2002). For example, this allowed consideration of the propensity of dabbling ducks to skim surface water (e.g. Pacific Black Duck), compared with waterbirds of the same family that graze (e.g. Australian Wood Duck, Plumed Whistling Duck) or forage in deep water (e.g. Black Swans) (Table 7.1).

The same analysis methods were used to investigate the effects of the percentage and availability of permanent water (km^2 per 1/4° grid), seasonal rainfall classification and interactions on Anseriformes abundance (log-transformed to remove heterogeneity of variance). The abundance of Anseriformes was estimated using reporting rate and bird-count data from Birds Australia (Barrett et al., 2003). Atlas data were collected during the 'New Altas of Australian Birds' project 1998–2002 from 279 000 bird surveys by 7000 observers. Australian Bird Count Data were collected during 79 000 surveys involving repeated counts of birds by 952 observers at 1681 sites between 1989 and 1995. Surveys for both datasets followed the standard methods for Birds Australia's 20-min, 2-ha search (Barrett et al., 2003), with the Australian Bird Count Data including complete counts of all individual birds observed, as well as the number of species observed. The relationship between the number of birds and number of species per observation was examined to test the use of reporting rate (number of surveys a bird species was present divided by the total number of surveys for each 1/4° map grid) as an index of abundance. The overall predicted Anseriformes abundance was then estimated separately for each 1/4° grid cell for Australia. Temporal data was not available for these analyses.

Analysis of LPAI surveillance data

Published sources of avian influenza wild-bird surveillance data were collated for Australia (n = 33 139 wild birds: Downie and Laver 1973; Downie et al., 1977; Mackenzie et al., 1984, 1985; Peroulis and O'Riley 2004; Hurt et al., 2006; Haynes et al., 2009) and overseas (n = 93 344 after Olsen et al., 2006). The first analysis used the global dataset to compare low pathogenic avian influenza (LPAI) prevalence between Australia and overseas and explored the effects of functional group, season and interactions. LPAI prevalence was estimated as the proportion of samples (cloacal, oropharyngeal or faecal swabs) from wild birds that were positive to low pathogenic Influenza A (via Polymerase chain reaction and / or virus isolations). Generalised linear mixed models and analysis of variance were again conducted using the *asreml* package (Butler et al., 2009) under R (R

Core Team, 2013) and site, sampling event and species were included as random variables. The second analysis used Australian avian influenza surveillance data to investigate the effects of bird species, functional group, log of the abundance of Anseriformes, the availability of permanent water, season, seasonal rainfall classification (a Bureau of Meteorology classification, identifying summer-dominant, summer, uniform, winter, winter-dominant, or arid rainfall) and interactions on the prevalence of LPAI in Australia, with site, sampling event and species included as random variables. For these two analyses bird species were aggregated into 14 functional groups (Figure 7.1). Stepwise regression was performed to remove non-significant terms (P>0.05). Wald tests were used to examine fixed effects and non-marginal terms not significant at the P = 0.05 level were dropped sequentially. Differences between terms for discrete variables are presented with 95% confidence intervals.

Comparison of sample methods: field trial

The estimates of prevalence of LPAI from cloacal, oropharyngeal and faecal samples were compared with quantitative real-time reverse transcriptase PCR (qRT–PCR) in a field trial in New South Wales where all three samples were collected from the same species at the same locations and sampling periods (n = 3242 samples from 2683 wild birds). These data are part of a larger surveillance dataset for eastern Australia (Hansbro et al., 2010), which were not included in the overall analysis of Australian surveillance data described above, but were used only to compare the three methods of sample collection. Transport media, storage, transport, operators, testing preparation and testing procedures were the same for all samples.

Details of the data-collection methods, sampling techniques and testing procedures are described elsewhere (Tracey, 2005; Kirkland and Tracey, 2006; see also Rose et al., 2006). Briefly, swabs were taken from live-captured or recently shot birds by inserting a swab deeply into the vent (cloacal) or oropharynx and swabbing the mucosa. The tip of the plastic-shafted swab was placed into a vial containing phosphate-buffered gelatin saline (PBGS) transport media (8 g of NaCl, 0.2 g of KCl, 1.44 g of Na_2HPO_4, 0.24 g of KH_2PO_4 dissolved in 800 mL of distilled H_2O). The viral transport medium was stored frozen, or at 4°C before use. Samples were maintained cold (4°C) throughout the transport process, and transported to the laboratory within 48 h of collection. Samples were either tested on delivery, or if not able to be completed within 48 h, were stored in a -80°C freezer (or -

20°C for serum samples). Testing was conducted at Elizabeth McArthur Agricultural Institute with qRT–PCR (cloacal, oropharyngeal and faecal) and the Influenza A group reactive competitive enzyme-linked immunosorbent assay (cELISA, serum), based on the method and reagents supplied by the Australian Animal Health Laboratory, Geelong (www.csiro.au/places/AAHL.html).

For faecal sampling, only freshly deposited moist samples were collected, the species or group of species were identified wherever possible, and a score given for the level of confidence in determining the species or group, as follows: Highly likely (sample collected immediately after a bird was observed defaecating), Likely (bird observed in the area immediately before collecting samples), Possible (bird observed in the area within 1 h of sampling), Unknown (birds known to occur in the area). The abundance of birds was estimated with point counts (Bibby et al., 2000) each morning before collecting samples at each site, which aided species identification. Size and shape of the faeces was distinguishable for different groups of species (ducks, large waders, small waders). The swab was lightly coated with faeces. Only samples where the species was identified as Likely or Highly likely were included when comparing sample techniques.

Costs of collection methods were estimated and included labour ($15 h–1), costs of consumables (feed for traps, ammunition), and the average number of samples collected per hour. To allow for direct comparison of collection methods, cost of travel (vehicle, fuel, labour) to sites was not included.

Risk profiles

Ecological and epidemiological information has been used to assign risks of exposure by wild-bird species and location according to a range of variables to achieve the following two main aims:

(1) to assess the risk of endemic LPAI viruses in wild birds becoming highly pathogenic through interactions with poultry (Surveillance Aim 1) and

(2) to assess the risk of wild birds introducing foreign subtypes of avian influenza (Surveillance Aim 2).

Risk profiles are consistent with 'exposure assessment' under the OIE risk analysis framework (Murray, 2002) and were developed for Surveillance Aim 1 by using (in order of importance): (a) log of poultry density (Robinson et al., 2007), (b) the estimated

prevalence of LPAI, and (c) the abundance of Anseriformes. Risk profiles developed for Surveillance Aim 2 used (in order of importance): (a) the probability of moving from areas where HPAI epizootics have occurred in 2003–09 (Food and Agriculture Organization of the United Nations, 2009; Office International Epizooties, 2009; World Health Organization, 2009), (b) the abundance of Anseriformes known to move into South-east Asia and (c) the estimated prevalence of LPAI.

To classify the risks to poultry, the risk of incursion was assumed to be dependent on poultry density (Robinson et al., 2007), using the natural log of the number of birds (Snow et al., 2007). The abundance of Anseriformes was estimated by using reporting rate and bird-count data from Birds Australia (Barrett et al., 2003). Atlas data were collected during the 'New Altas of Australian Birds' project 1998–2002 from 279 000 bird surveys by 7000 observers. Australian Bird Count Data were collected during 79 000 surveys involving repeated counts of birds by 952 observers at 1681 sites between 1989 and 1995. Surveys for both datasets followed the standard methods for Birds Australia's 20-min, 2-ha search (Barrett et al., 2003), with the Australian Bird Count Data including complete counts of all individual birds observed, as well as the number of species observed. The relationship between the number of birds and number of species per observation was examined to test the use of reporting rate (number of surveys a bird species was present divided by the total number of surveys for each 1/4° map grid) as an index of abundance. For each 1/4° grid cell, abundance and prevalence was estimated separately for functional groups, which was found to be important in predicting LPAI prevalence (see Results).

Distances moved and movement probabilities for Anseriformes were estimated using banding data from the Australian Bird and Bat Banding Scheme (www.environment.gov.au/biodiversity/science/abbbs/, accessed 1 January 2009). Spatial analyses were conducted in Arcview 3.2 (ESRI, Redlands, CA) and Manifold® (Carson City, NV). In addressing Surveillance Aim 2, movement probabilities were estimated for species identified as conducting regular or occasion movements in South-east Asia (after Tracey et al., 2004; Delaney and Scott, 2006; Table 6.1), using a movement probability model (see Results; $y = 22\,928\,x^{-2.2541}$, where x is the distance to the current distribution of HPAI epizootics).

Prevalence of LPAI for each grid cell (p total) was estimated by

$$p_{total} = \frac{\sum (p_f \times a_f)}{\sum a_f}$$

where p_f is the prevalence of LPAI according to the functional group, using Australian surveillance data (Table 7.1) and a_f is the abundance index for Anseriformes in each functional group in each 1/4° grid cell.

The final scores were calculated with a normalised weight (w_i), by using a rank sum (2) (Malczewski, 1999),

$$w_i = \frac{n - r_j + 1}{\sum (n - r_k + 1)}$$

where *wi* was the normalised weight for the *j*th criterion, n was the number of criteria under consideration (k = 1, 2, ..., n), and r_j was the rank position of the criterion. Each criterion was weighted (n - r_j + 1) and then normalised by the sum of all weights, i.e. $\sum(n$ - r_k + 1).

The value for each criterion for each grid cell was normalised by using (3) before applying weights, as follows:

$$\delta = \frac{d - d^{\min}}{d^{\max} - d^{\min}}$$

where δ is the normalised value and d is the original value.

Table 7.1. Prevalence of low-pathogenicity avian influenza (LPAI) of Australian Anseriformes

[a]Superscripts (after Christidis and Boles 2008): V = vagrant to Australia (fewer than 10 records); I = introduced to Australia; AAT = Australian Antarctic Territory; LH = Lord Howe Island; u = subfamily unresolved, based on Livezey (1986); Sraml *et al.,* (1996); Johnson and Sorenson (1999). [b]estimated for functional groups (after Roshier *et al.,* 2002). [c]after Tracey *et al.,* 2004; Delaney and Scott 2006.

Common Name[a]	Scientific Name	Functional Group	Prev (%) of LPAI[b]	Movements into SE Asia[c]
Anseranatidae				
Magpie Goose	*Anseranas semipalmata*	Dabbling ducks	3.12	Regular
Anatidae				
Anatinae (dabbling ducks)				
Australian Wood Duck[U]	*Chenonetta jubata*	Grazing waterfowl	0.71	Unknown
Cotton Pygmy-goose[U]	*Nettapus coromandelianus*	Dabbling ducks	3.12	Occasional
Green Pygmy-goose[U]	*Nettapus pulchellus*	Dabbling ducks	3.12	Occasional
Garganey	*Anas querquedula*	Dabbling ducks	3.12	Rare
Australasian Shoveler	*Anas rhynchotis*	Dabbling ducks	3.12	Unknown
Northern Shoveler[V]	*Anas clypeate*	Dabbling ducks	3.12	Rare
Grey Teal	*Anas gracilis*	Dabbling ducks	3.12	Occasional
Chestnut Teal	*Anas castanea*	Dabbling ducks	3.12	Unknown
Northern Pintail[V]	*Anas acuta*	Dabbling ducks	3.12	Rare
Kerguelen Pintail[AAT/V]	*Anas eatoni*	Dabbling ducks	3.12	Unknown
Mallard[I]	*Anas platyrhynchos*	Dabbling ducks	3.12	Unknown
Pacific Black Duck	*Anas superciliosa*	Dabbling ducks	3.12	Occasional
Anserinae (swans and geese)				
Cape Barren Goose[U]	*Cereopsis novaehollandiae*	Grazing waterfowl	0.71	Unknown
Black Swan	*Cygnus atratus*	Deep-water foragers	0.94	Unknown
Mute Swan[I]	*Cygnus olor*	Deep-water foragers	0.94	Unknown
Canada Goose[V/I]	*Branta Canadensis*	Grazing waterfowl	0.71	Unknown
Aythyinae (diving ducks)				
Hardhead	*Aythya australis*	Deep-water foragers	0.94	Unknown
Dendrocygninae (whistling ducks)				
Spotted Whistling-Duck	*Dendrocygna guttata*	Grazing waterfowl	0.71	Rare
Plumed Whistling-Duck	*Dendrocygna eytoni*	Grazing waterfowl	0.71	Rare
Wandering Whistling-Duck	*Dendrocygna arcuata*	Dabbling ducks	1	Regular
Oxyurinae (stiff-tailed ducks)				
Musk Duck	*Biziura lobata*	Deep-water foragers	0.94	Unknown
Blue-billed Duck	*Oxyura australis*	Deep-water foragers	0.94	Unknown
Stictonettinae (freckled duck)				
Freckled Duck	*Stictonetta naevosa*	Dabbling ducks	3.12	Unknown
Tadorninae (shelducks)				
Radjah Shelduck	*Tadorna radjah*	Dabbling ducks	3.12	Rare
Australian Shelduck	*Tadorna tadornoides*	Grazing waterfowl	0.71	Unknown
Paradise Shelduck[LH/V]	*Tadorna variegata*	Grazing waterfowl	0.71	Unknown
Pink-eared Duck	*Malacorhynchus membranaceus*	Dabbling ducks	3.12	Unknown

7.3 Results

Analysis of Anseriformes movements and abundance

The number of species recorded during Australian Bird Counts was found to be sufficient in predicting the log of the number of birds per observation ($y = 0.6913x + 1.4456$, $P < 0.001$). Movement probabilities were estimated for Anseriformes by using distance moved from recapture data ($n = 8095$), with a power model showing a good fit to the data ($r^2 = 0.89$, $y = 22\ 928x^{-2.2541}$). From banding and recovery data, within 14 days of capture, 75% of birds remained within 5 km of capture, 80% within 10 km, 90% within 35 km, and 95% within 100 km; the maximum distance moved from the capture location was 2305 km (n= 1314). Seasonal rainfall classification (Wald statistic $F_{(6, 348)} = 6.129$, P<0.0001) and mean annual rainfall (Wald Statistic $F_{(1,, 443)} = 9.976$, P=0.0017) was important in predicting the movements of Anseriformes (Appendix III), with greater movements in the summer dominant rainfall areas of northern Australia.

The abundance of Anseriformes was correlated with the availability of permanent water (Wald statistic $F_{(1,\ 13\ 424)} = 265.8$, $P < 0.0001$, $y = 28.248x + 0.448$, $r^2 = 0.814$) and seasonal rainfall classification (Wald statistic $F_{(5, 13\ 424)} = 55.8$, $P < 0.0001$), as follows (in a decreasing order of abundance – mean reporting rate): uniform (0.57 ± 0.05, n=1 291), summer (0.47 ± 0.03, n=1 840), winter-dominant (0.46 ± 0.05, n=496), winter (0.42 ± 0.02, n=2 631), summer-dominant (0.29 ± 0.03, n=2625) and arid (0.19 ± 0.02, n=4 548) rainfall zone.

Analysis of Australian surveillance data

LPAI prevalence was significantly greater in dabbling ducks than in all other functional groups for Australia and overseas (Wald statistic $F_{(14, 208)} = 3.236$, P <0.0001, Appendix III, Figure 7.1). Prevalence of LPAI was significantly lower in Australia (1.04% ± 0.06, n=29,167) than that in other countries (4.67% ± 0.02, n=95,441) (Wald statistic $F_{(1, 174)} = 8.630$, $P < 0.003$, Appendix III), with the prevalence 2.6–4 times less for all functional groups, with the exception of seabirds, where there was no significant difference between Australia and overseas, and small migratory waders, where the prevalence was 11 times less in Australia than in overseas (Figure 7.1). However, sampling is unlikely to be representative across all species and locations, particularly for dabbling ducks in North America where many samples are regularly taken in areas of previous high virus activity.

In Australia and overseas, LPAI was either not detected or was of low prevalence for other functional groups (quail and wild Galliformes: 0/27, 4/899 (Australia, overseas); large waders: 0/58, 0/87; small resident waders: 0/260, 1/58; birds of prey: 0/6, 2/192; pigeons and doves: 0/1, 1/166; or bush birds: 0/34, 0/92).

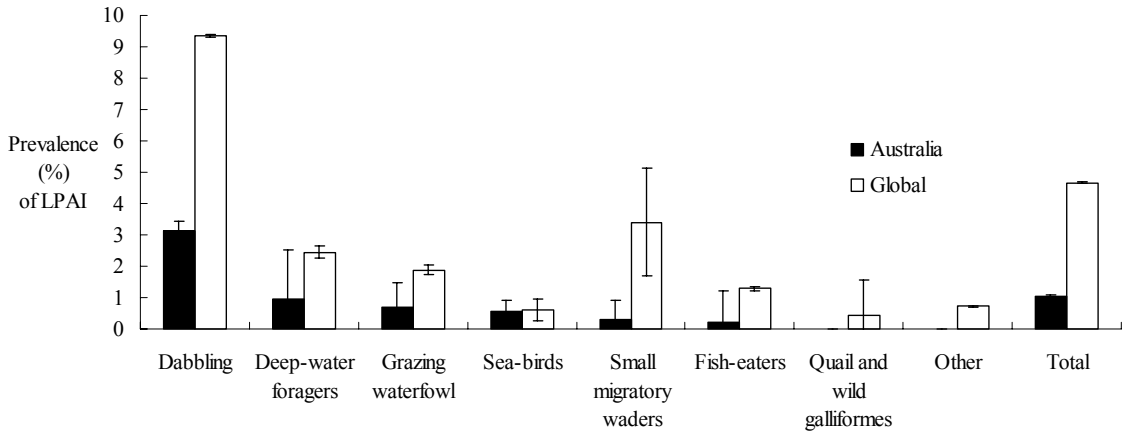


Figure 7.1. Prevalence (% with s.e.) of low-pathogenicity avian influenza (LPAI) in wild birds by functional group from Australian (black) and global (white) surveillance data. Australia: n=29 167; Global: n=93 344 (after Olsen et al., 2006).

Comparison of sample methods: field trial

The cost of collecting faecal samples ($1.95 per sample) was less than the cost of collecting samples by shooting ($6.12 per bird) and trapping ($9.10 per bird). When compared on the same populations during the same time periods, detection of antibodies with cELISA from serum (18.45% ± 4.38, 95% confidence interval, n=374) was much more likely than detection of the virus (1.07% ± 2.16, n=2 868). Also, the prevalence of LPAI viruses detected with qRT–PCR was similar for cloacal (2.27% ± 0.97, n=948) and oropharyngeal (2.17% ± 2.3, n=185) samples, whereas it was significantly lower from faecal samples (0.29% ± 0.28, n=1 735).

Risk profiles

On the basis of risk profiles, highest priorities to assess risks of endemic viruses becoming highly pathogenic are in the region of state capitals, i.e. Melbourne, Sydney, Brisbane, Adelaide, Perth, Darwin and Hobart, and in the Mareeba area near Cairns, Queensland (Figure 7.2a). Current poultry densities for areas where previous HPAI epizootics occurred are 266 km^{-2} for Keysborough, Victoria (1976, Turner, 1976), 464 km^{-2} for Bendigo, Victoria (1985, Barr et al., 1986; 1992, Selleck et al., 1997), 222 km^{-2} for Lowood,

Queensland (1994, Westbury 1998), and 900 km^{-2} for Tamworth, New South Wales (1997, Selleck et al., 2003). Highest priorities to assess risks of wild birds introducing foreign viruses are the regions of north-western Australia from Broome through to Arnhem Land, particularly in the Kimberley, Western Australia (Figure 7.2b). Combined ranks to address both surveillance aims include all these high-priority locations (Figure 7.2c). Australian surveillance (1971–2007) has generally occurred in these priority areas, with the exception of Brisbane, Darwin and Mareeba, where surveillance is currently underway.

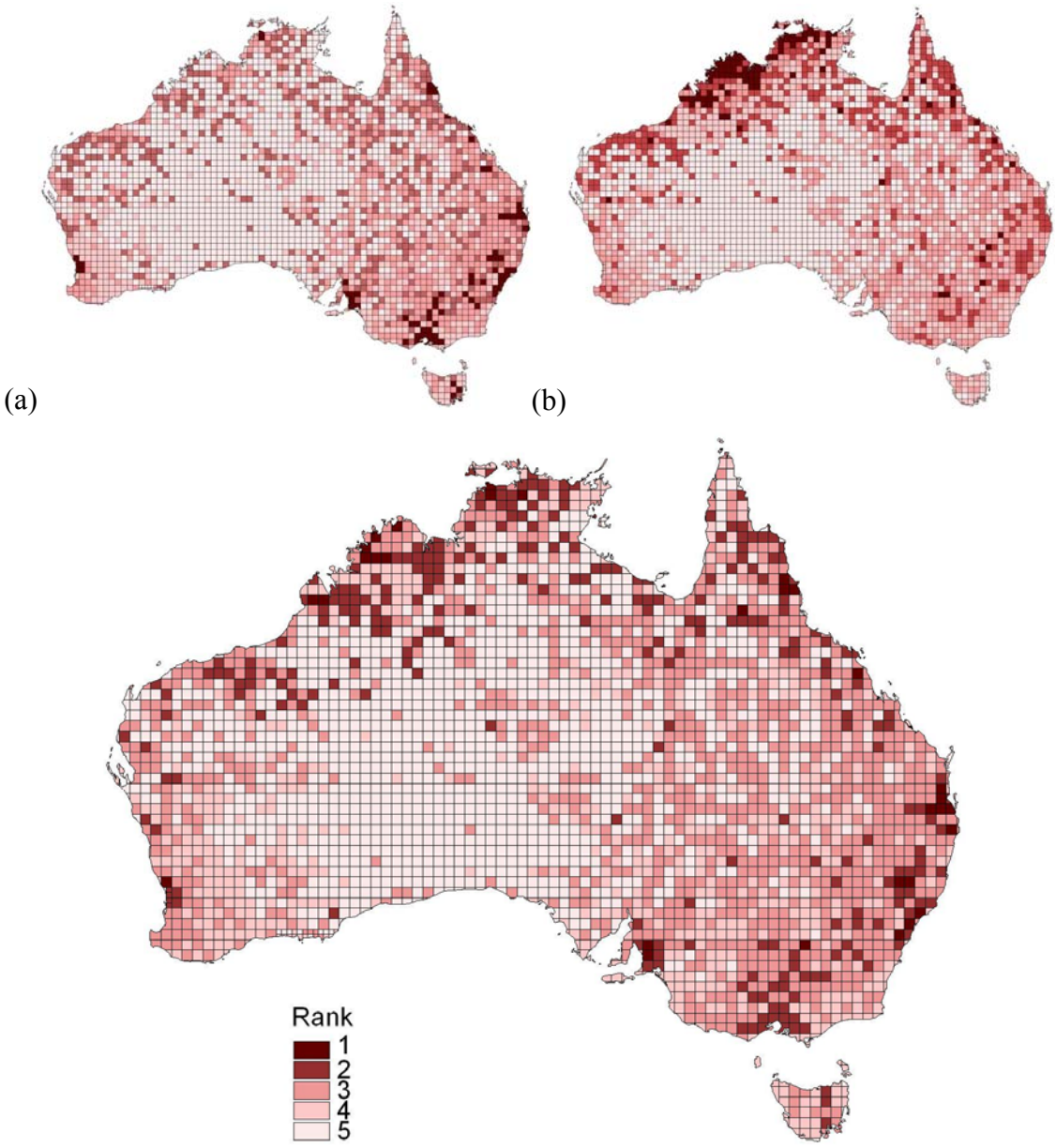


Figure 7.2. Priorities for the surveillance of avian influenza in Australia's wild birds: (a) to assess the risk of endemic low-pathogenicity avian influenza viruses in wild birds

becoming highly pathogenic through interactions with poultry (Surveillance Aim 1), (b) to assess the risk of wild birds introducing foreign subtypes of avian influenza (Surveillance Aim 2), and (c) a combined classification to address Surveillance Aims 1 and 2. Priorities (Rank 1 (highest) to 5 (lowest)) are based on risk profiles developed using the log of poultry density (Robinson et al., 2007), the estimated prevalence of low-pathogenicity avian influenza by functional group using Australian surveillance data (Downie and Laver, 1973; Mackenzie et al., 1984; Mackenzie et al., 1985; Peroulis and O'Riley, 2004; Hurt et al., 2006; Haynes et al., 2009), the abundance of Anseriformes (source: Birds Australia), and the probability of Anseriformes moving from areas where HPAI epizootics have occurred in 2003–09 (FAO 2009; OIE 2009; WHO 2009; source: Australian Bird and Bat Banding Scheme).

7.4 Discussion

Anseriformes and avian influenza in Australia

Abundance and movement patterns for Anseriformes are found to be irregular, varying with the availability of permanent water and seasonal rainfall, which is commonly reported for Australian anatids, with flood events and temporary rainfall particularly important (Roshier et al., 2001a, b). Anatids are often more dispersive in arid areas, and more sedentary and abundant on permanent water (Frith, 1982; Woodall, 1985). The greater Anseriformes movements in the summer dominant rainfall areas is likely a consequence of the distinct differences between wet and dry seasons of northern Australia, where most rainfall occurs in 3 months of the year. Large numbers of waterbirds can congregate during the late dry season (May–October) on persistent swamps and disperse very widely during the wet (November–April) (Morton et al., 1990a). However, cues for and patterns of anatid movements are complex, determined by individual behavioural strategies (Roshier et al., 2008) as well as surface water over large spatial scales (Roshier et al., 2001a).

The abundance and movements of waterbirds have implications for understanding persistence of LPAI in natural reservoirs and for managing HPAI epizootics. In Australia, LPAI would be expected to peak in Anseriformes during 'boom' breeding periods where thousands of birds congregate on major wetlands or floodplains. This occurs infrequently, with the largest breeding events occurring perhaps once every 10 years. In tropical Australia, LPAI may be more seasonal with peaks expected in the late dry season, and

greater potential for dispersal of LPAI during the wet. This is consistent with HPAI H5N1 in tropical South-east Asia, where prevalence was significantly higher when large flocks of Anseriformes congregate during the dry season (=southern winter) (Siengsanan et al., 2009).

The lower prevalence of AI found in Australia than overseas (Olsen et al., 2006; Haynes et al., 2009); is likely to be a result of differences in the behaviour and movements of Australian Anseriformes from those overseas (Tracey et al., 2004; McCallum et al., 2008). Australia is dry with irregular rainfall and as a consequence breeding and movements of waterbirds are irregular. During the past 20 years, breeding has been infrequent and waterbird abundance has declined markedly in some areas (Porter et al., 2006), by up to 80% for some species (Nebel et al., 2008). Loss of wetlands because of dams, water extractions and levee banks, particularly in south-eastern Australia, is likely to have contributed to these declines (Kingsford 2000; Nebel et al., 2008).

The persistence of avian influenza viruses is likely to be affected by the regularity of breeding, as well as movement patterns, both being correlated with water availability (Frith 1982). Hence, breeding occurs in southern Australia in spring and in northern Australia at the end of the wet season (April–May, southern autumn). During severe drought, most Australian anatids do not breed (Frith, 1982), which is likely to limit LPAI prevalence. Increased virus prevalence following breeding is often observed or assumed for animal pathogens, including avian influenza virus (Hinshaw et al., 1985; Alfonso et al., 1995), as a result of the boost in immunologically naïve individuals (juveniles) (Clark and Hall, 2006; Munster and Fouchier, 2009).

LPAI in Australian wild birds was highly variable among sampling periods and locations and no seasonal trends were apparent. This is likely to be a consequence of a lack of long-term studies (low sample sizes over time) coupled with a high variability in rainfall and Anseriformes movements and abundance between climatic zones (northern: wet season – summer-dominant rainfall v. southern: uniform or winter-dominant rainfall).

Functional group was clearly important in predicting LPAI prevalence, with dabbling ducks identified as the main reservoir for Australia and overseas. The propensity of dabbling ducks to skim surface water is a likely explanation (Olsen et al., 2006). Avian

influenza viruses are known to persist in water (Webster et al., 1978; Stallknecht et al., 1990a, b; Brown et al., 2009; Roche et al., 2009) and high levels of faecal material may occur on the surface (Lang et al., 2008).

The substantial difference in prevalence (11 times) for small migratory waders between Australia and overseas may suggest that these species are unlikely to be responsible for transferring viruses into Australia. Rather, these species may act as a sentinel for endemic viruses maintained by dabbling ducks. Seabirds also conduct regular global travel but, in contrast, have similar prevalence between Australia and overseas. This may support the view that seabirds maintain viruses that are unique from viruses on mainland Australia, which is consistent with their behaviour and movements and phylogenetic differences among virus groups (Munster and Fouchier, 2009). Future investigations of genetic differences between Australian and Eurasian and American subtypes may confirm these trends.

Practical considerations of sample-collection methods and testing procedures are important to consider when interpreting results of surveillance (Munster et al., 2009) and in preparation for future HPAI epizootics. The higher sero-prevalence (18.45% ± 4.38, 95% confidence interval, n=374) cf. virus prevalence (1.07% ± 2.16, n=2 868) reported here is typical of disease studies, with avian influenza virus normally detectable from swabs for up to 5 days, after which antibodies may then be detected for considerable periods, e.g. up to 12 months, depending on dose at exposure (Calnek 1997). Although variable, faecal sampling was three and five times less expensive than sampling involving shooting and trapping respectively. However, the significantly lower prevalence from faecal samples than from cloacal samples highlights the need for reporting results separately. Possible reasons include degradation of samples (low volumes of RNA), or contamination as a result of excess faecal material or other substances from the environment. In comparison, Pannwitz et al., (2009) reported similar recovery rates from faecal and cloacal samples for some species (geese and swan, but not ducks). Pannwitz et al., (2009), however, compared recovery rates from different locations and time periods, which is problematic because of low prevalence and considerable variation in prevalence commonly reported between locations and over time. Improved collection procedures for faecal samples may increase the rate of detection, for example, by minimising the amount of faecal material, or collecting samples from hardened or more sterile surfaces (e.g. concrete, bitumen,

compacted soil or gravel, sand and decks). The prevalence from cloacal swabs was not significantly different from that from oropharyngeal swabs, which is consistent with Peroulis and O'Riley (2004). However, Ellström et al., (2008) and Munster et al., (2009) reported significantly higher LPAI prevalence from cloacal samples. In comparison, for HPAI H5N1, virus recovery was significantly higher from the respiratory tract than from the cloaca (Sturm-Ramirez et al., 2005; Keawcharoen et al., 2008). For on-going surveillance, faecal (environmental) samples may be collected as a rapid and cost-effective means of investigating virus presence. However, to verify virus prevalence, the collection of oropharyngeal and cloacal samples from hunted or captured birds is recommended.

Risk profiles

Risks posed by highly pathogenic avian influenza to poultry and humans and associated control measures have been considered by many authors (Alexander, 1993; Tracey et al., 2004; Perdue and Swayne, 2005; Animal Health Australia 2011; Chapter 6). Of interest here is to improve the efficiency and relevance of low pathogenic avian influenza surveillance in wild birds, which is an important first step to understanding the ecology of avain influenza viruses.

There is some uncertainty as to the role of poultry density in initiating HPAI in Australia. Although a shift in pathogenicity for avian influenza can occur rapidly (Brugh and Beck, 1992; one or two passages; Arzey, 2005), population size or density is likely to be important in determining the levels of prevalence, transmissibility and mutation rates for many viruses, e.g. rabbit haemorrhagic disease virus in rabbits (Calvete and Estrada, 2000; Henzell et al., 2002), brucellosis in bison (Dobson and Meagher, 1996), *Mycoplasma gallisepticum* in house sparrows (Hochachka and Dhondt, 2000), including avian influenza (Bunn, 2004; Turner, 2004; Pfeiffer et al., 2007; Snow et al., 2007). Westbury (1998) suggested that poultry-farm density was low in the first four HPAI epizootics in Australia. However, the current study indicates that both poultry-farm density and poultry density are highest in the areas where previous epizootics occurred; areas where all five HPAI epizootics took place are ranked highest by using poultry population per 1/4° grid. Hamilton et al., (2009) identified the density of poultry farms as a risk factor for HPAI in Australia, listing five regions (the Sydney region, Central Coast NSW, Tamworth, Mornington Peninsula and Bendigo) that had poultry-farm density equal to or greater than regions of Canada and Italy affected by large epizootics of HPAI (>0.05 farms km^{-2}),

which is consistent with the current study. Hamilton et al., (2009) also emphasised the importance of biosecurity measures to prevent the spread of the virus from infected farms in the event of an epizootic, resulting from service providers regularly contacting multiple farms.

Although currently unavailable, future risk profiles could incorporate additional variables, including housing (caged, floor, free range, barn, deep litter; Pfeiffer 2006; Fossum et al., 2009) and the type of operation (pullets, breeders, broilers, layers; Snow et al., 2007). Poultry-farm density may also be more appropriate than poultry density in predicting spread, once an epizootic occurs (Truscott et al., 2007; Hamilton et al., 2009).

There has been considerable debate on the ability of wild birds to spread HPAI virus over large distances while infectious (up to 14 days, Kida et al., 1980), with several recent studies suggesting that this is likely to have occurred (Sabirovic et al., 2006; Stallknecht and Brown 2008). However, the persistence of virus in the environment, the connectivity of the landscape relevant to wild birds (particularly dabbling ducks, Roshier et al., 2001a) and the frequency of movements for multiple species (McCallum et al., 2008) are likely to be more important than individual bird movements within short periods. Bird populations can maintain avian influenza viruses despite low prevalence (Stallknecht and Brown 2008) and viruses can remain infective in freshwater lakes for 4 days at 22°C, more than 30 days at 0°C (Webster et al., 1978), or up to 200 days at 17°C, when virus concentrations are higher (Stallknecht et al., 1990b).

When estimating the distance moved from banding data, there are several biases that should be considered when interpreting risk profiles. In particular, individual ducks are more likely to be recaptured at the same location than elsewhere when consecutive trapping periods occur at the same location. This would create an underestimate of HPAI risk for these criteria. Recoveries may also be more likely where damage mitigation permits are issued to protect rice. Satellite transmitters have demonstrated that large movements of grey teal can occur within hours (up to 345 km) (Roshier et al., 2006), and within days (up to 1268 km) (Roshier et al., 2008), with some birds returning to their point of origin. These individual movements would have been difficult to detect with banding studies. However, movement probabilities estimated in the current study ($y = 22\ 928x^{-2.2541}$, see Results) are consistent with overall patterns of movement reported using satellite

transmitters (Roshier et al., 2006, 2008). For example, Roshier et al., (2006) found that 78% and 83% of grey teal movements occurred within 5 km in the Riverina and Lake Eyre Basin respectively.

To develop risk profiles for avian influenza in wild birds, a range of simple seasonal and climatic variables have been explored to explain the abundance of anatids and the likelihood of their movement over a large area. However, these ignore the finer-scale processes of wetland quality, the temporary availability of wetlands and flood events. These are known to be important in predicting anatid movements and abundance, particularly in arid Australia; however, they are difficult to incorporate when presenting spatial data that can be interpreted over time.

Although highest priorities have been assigned to dabbling ducks, because they represent the major reservoir of LPAI in Australia, surveillance of other species should not be excluded. Migratory and resident Charadriiformes, seabirds (including pelagic gulls and terns and Procellariiformes), quail, ratites and other functional groups may also play a role in maintaining avian influenza viruses, including those with unique lineages (e.g. gulls and terns, Munster and Fouchier 2009).

There are many uncertainties that affect the risks of an incursion of HPAI. Risk profiles developed here are not for predicting future epizootics, but rather, are a tool to maximise the efficiency and relevance of wild-bird surveillance, and to provide insights into patterns of LPAI occurrence. Hence, poultry producers should continue to maintain high biosecurity (including limiting contact with wild birds, regular treatment of water, rapid reporting of unusual mortalities), regardless of whether they are located in high- or low-priority areas. The major risks for poultry operations are likely to be Anseriformes in the vicinity, a failure in biosecurity (e.g. water quality or entry of contaminated personnel) and confined poultry of sufficient density to allow development and dissemination of a pathogenic virus (Bunn 2004). However, there are other potential sources of LPAI, including live-bird markets and movements of domestic birds (poultry, turkeys, ducks, emus, quails) (Arzey 2004), and more important modes of transmission following outbreaks of HPAI (Sims et al., 2005; Feare 2007; Gilbert et al., 2008; Hamilton et al., 2009). In Australia, service providers regularly contact multiple farms (Hamilton et al., 2009) and are a direct potential source of secondary spread.

Previous surveillance for avian influenza in Australia has generally occurred in areas identified as highest priority, with the exception of Mareeba (northern Queensland), Brisbane and Darwin, with the current Avian Influenza Wild Bird Surveillance Program addressing these gaps. This surveillance has provided valuable information on the role of wild birds in maintaining LPAI viruses, and provides the basis for future insights into global patterns of avian influenza, in particular in the investigation of genetic similarities of subtypes between continents. However, surveillance has been sporadic, with a limited number of samples collected (35,000 samples in 1970–2007 in Australia v. 300,000 samples per year in other countries, Munster and Fouchier 2009) and with information on bird abundance, age, behaviour, breeding and movements rarely being collected during surveillance activities. This limits our ability to offer explanations for the spatial and temporal variability of virus prevalence. Enhanced surveillance in priority areas that incorporates ecological information over a longer time frame is important to validate trends of LPAI prevalence, in understanding the main determinants for virus spread and persistence, and in predicting and managing future epizootics of HPAI in Australia.

8. GENERAL DISCUSSION

Invasive species cause significant economic, environmental and health-related impacts world-wide, and rates of invasion are increasing (Genovesi et al., 2009; Hulme, 2009). In the United States invasive species cost almost $120 billion per year, and are the primary threat to 42% of threatened and endangered species (Pimentel et al., 2005). Similarly in Australia, annual economic, environmental and social costs of pest animals ($1 billion: McLeod, 2004; Tracey et al., 2007) and weeds ($4 billion: Sinden et al., 2004) are substantial. Diseases of invasive species can also significantly impact upon economies (Bennett et al., 2009), the environment (Daszak et al., 2000) and human (Binder et al., 1999) and animal (Gortázar et al., 2007) health. Disease events are increasing in frequency and most originate in wildlife (Jones et al., 2008). Quantifying these impacts and thorough evaluations of cost-effective management strategies are essential in dealing with the on-going threats of invasive species.

In this thesis I used case studies of pest birds to develop improved, efficient methods to estimate pest impact and disease risk, and evaluate strategies to manage these impacts, in agricultural crops, on native fauna and in relation to disease risk. As opposed to our knowledge of pest mammals (e.g. Lever, 1985; Putman, 1989), there are fundamental deficiencies in our knowledge of pest birds and their role in maintaining viruses, their impacts and how to measure them, and the costs and efficacy of commonly used management practices.

Pest bird populations can increase rapidly, causing significant and localised impacts. Bird populations are particularly difficult to manage, often with many different species involved, high densities, high mobility, varied diet (Hasebe and Franklin, 2004) and rapid population fluctuations with changes to habitat, climatic conditions, food availability, nest sites and broad scale changes to agricultural practices (Chamberlain et al., 2000; Higgins et al., 2006; Rintala and Tiainen, 2008). These latter factors are often more important than direct interventions such as lethal controls in regulating populations and associated impacts (Murton et al., 1974) and without taking these into consideration, measures to control bird populations usually fail (Feare, 1991).

Pest species ecology and population dynamics will determine short term and long term impacts and efficacy of management strategies (Dolbeer, 1998; Hone, 2007). In my study, one-hundred and forty bird species were observed of which twenty-nine Australian natives and seven introduced species are known to damage fruit (Tracey et al., 2007). Their abundance and impacts varied significantly in time and space. This suite of species also have major differences in ecology, behaviour and movements, and are therefore unlikely to respond to management techniques in the same way.

There are few established techniques available for measuring *economic impacts* of birds (DeHaven and Hothem, 1979; Nemtzov, 2004) and these are generally time-consuming, or can be unreliable and inaccurate. I developed a new method for estimating bird damage in wine grapes, which improved sampling efficiency by over 65% compared to other methods and has application to most crop-bird situations.

Current strategies used for managing birds have rarely been rigorously evaluated in terms of their ability to reduce abundance or impact. Due to high variability in bird-crop systems, bird species, abundance (Dyer, 1967; Tracey et al., 2001; Tracey et al., 2007), and the extent of damage (Wiens and Dyer, 1977; Whitehead et al., 1995), large sample sizes are required to confidently assess management treatments (Tracey et al., 2001). Empirical studies with sufficient sample sizes had been lacking for even the most commonly used techniques for managing birds, including shooting, trapping, netting and acoustic and visual deterrents.

In my study, generalised linear mixed models were used to test the effects of netting, shooting and scaring treatments on percent bird damage across 185 property year records. Netting was the most effective treatment, despite birds regularly breaching nets. Shooting was not as effective but was one-third of the cost and had 13% lower damage compared with nil-treatments. Scaring with electronic devices and visual deterrents had no effect on bird damage in fruit crops. Despite their widespread use, lethal methods had limited effectiveness for reducing pest populations in vineyards and orchards. However, shooting to scare birds from the crop, rather than to control populations was effective in reducing damage. In addition, systematic searches indicate that nest removal can significantly reduce starling reproduction and could have benefits for the long-term management of

starling populations. These findings have significant implications for producers in cost-effectively managing the annual impacts of pest birds.

While this thesis provides valuable information on the immediate effects of bird management, compensation and changes to bird populations over longer time frames were not considered. Further study is recommended to evaluate the timing for control and the carryover effects of management from one period to the next. Studies are required that measure compensation and incorporate natural changes in populations (growth and natural mortality) and timing of control (recruitment vs juvenile mortality) when evaluating management. This is particularly relevant for species that have a high rate of replacement and population turn-over (high rates of recruitment and mortality), such as starlings and mynas. For example any bird that escapes control in one period has the potential to reproduce and thus increase populations in later periods. When considering costs and benefits of control in this way a dynamic bio-economic framework can be developed and used (Clark, 1990; Ellner and Guckenheimer, 2006). This allows a change in the assumption of profit maximisation for pest bird control for a single season or year to an assumption of maximising returns over a longer period.

Invasive species can cause significant *environmental impacts* leading to extinctions (Clavero and García-Berthou, 2005), and affecting ecosystem function (O'Dowd et al., 2003; Sanders et al., 2003). While biodiversity impacts of pest animals are mainly attributed to mammalian predators (Coutts–Smith et al., 2007), pest birds also compete (Stone, 1996; Grarock et al., 2012) and hybridise (Williams and Basse, 2006) with native species. Impacts of pests on the environment are difficult to separate from other causes of habitat loss or disturbance (MacDougall and Turkington, 2005) and can occur over long time frames (Davis 2003). Establishing evidence of environmental impacts based on cause and effect (rather than correlations in species abundance) (e.g. Conroy et al., 1989) is difficult, and rarely conducted for pest birds.

When considering environmental impacts on Lord Howe Island, phenotypic characteristics suggested that mallards have supplanted the native Pacific black duck, with 81% of birds classified as mallard or mallard-like hybrids, 17% as intermediate hybrids and only 2% as Pacific black duck-like hybrids. No pure Pacific black duck were observed. While these hybrids pose direct impacts to Pacific black duck, their indirect economic, social and

environmental impacts and their ecological role on Lord Howe Island requires further consideration. A management program to remove mallards using trapping, shooting and opportunistic capture by hand was conducted, with hand capture as the most cost-efficient technique (AUS$3.50/bird) followed by trapping (AUS$19/bird) and shooting (AUS$22/bird). Standardised indices of duck abundance before and after management indicate that the total population was reduced by over 70%. An understanding of the impacts and movements of mallards is essential in deciding the appropriate management action. If warranted the eradication of hybrid ducks on Lord Howe Island may be feasible if movements and reintroductions are infrequent and therefore able to be controlled.

These findings demonstrate that some introduced species can significantly alter the genetic integrity of native fauna, leading to their extinction. While mallards are non- indigenous and widespread in the South Pacific and other parts of the world their impacts on native fauna are often unknown and rarely managed (Guay and Tracey 2009). Estimating impacts using phenotypic characteristics can also underestimate the extent of hybridisation (Braithwaite and Miller 1975; Green et al. 2000; Kulikova et al. 2004). It is also important also to consider the impacts of mallards and hybridisation in a broader ecological context (e.g. Hone 2007). Will a hybrid fulfil an ecological function equivalent to native precedcesor? Further study of the environmental impacts and management of mallards are recommended; including improved information on their indirect impacts, their distribution and abundance, and the use of genetic markers to verify the extent of hybridisation with native Anatidae.

Health-related impacts of invasive species can be significant and high on the community agenda. Emerging disease events are increasing as a result of a range of socio-economic, environmental and ecological factors, with wildlife the source for the majority of these (Jones et al., 2008). Ten to 20 influenza pandemics have occurred since the 1700's, with the 'Spanish flu' causing deaths to more than 20 million people, with wild aquatic birds the most likely source (Webster 1998).

Estimating the impact of diseases involves an understanding of transmission and the effect of the pathogen on the host (McCallum 2000). Preventative measures will require an understanding of the likelihood and risks posed by future disease outbreaks. The

epidemiology of viruses in wildlife and their interaction with humans and livestock are important in assessing their risks.

There has been a significant increase in surveillance for diseases in wildlife, particularly in wild birds. However, broad-scale surveillance is logistically difficult and costly because of natural low prevalence of viruses, and wide variety and abundance of potential hosts. Improvements are needed in targeting surveillance according to associated risks, and to improve efficiency.

In this thesis health-related impacts of birds were considered using avian influenza in Australasia as a case study. Since the first cases of H5N1 highly pathogenic avian influenza (HPAI) in humans in 1997 in Hong Kong (Xu et al., 1999), avian influenza has become internationally recognised by the public health practitioners, the animal health community and the general public. This virus is considered the precursor to subsequent major epizootics in 2001-2 (Guan et al., 2004) and 2003-4 (Li et al., 2004). By March 2004 epizootics were confirmed in China, Cambodia, Indonesia, Japan, Laos, South Korea, Taiwan, Thailand and Vietnam. Epizootics of H5N1 in poultry have since occurred throughout Asia, Europe and Africa, and its eradication is considered unlikely (Li et al., 2004; Sims and Narrod, 2009). H5N1 has also caused disease and death in humans (Claas et al., 1998; Subbarao et al., 1998; Yuen et al., 1998) via avian-to-human transmission.

The potential transmission of the H5N1, and other influenza A viruses from Asia to Australia via wild birds is of concern. There are many bird species known to undertake movements between Asia and Australia; the species involved, their movement behaviour, ecology and susceptibility to disease are all of importance when assessing the risks of avian influenza in Australia. I undertook detailed analysis of the abundance, movements and breeding ecology of Australia's Anseriformes in relation to the prevalence of low-pathogenic avian influenza (LPAI) and provide risk profiles to improve the efficiency and relevance of wild-bird surveillance.

Analysis of Australian wild-bird surveillance data strongly supports other studies that have found the prevalence of LPAI in wild birds to be much lower (1%) in Australia than that in other countries (4.7%). LPAI prevalence was highly variable among sampling periods and locations and significantly higher in dabbling ducks than in other functional groups. Trends

in Anseriformes movements, abundance and breeding are also variable, and correlated with rainfall, which could explain low prevalence and the failure to detect seasonal differences in LPAI in wild birds.

Risk profiles were developed to assess risks and to improve the efficiency and relevance of wild bird surveillance. These estimate risks based on likely interactions with poultry, poultry density, estimated LPAI prevalence, the abundance of Anseriformes, and the probability of Anseriformes moving from areas of HPAI epizootics. Overall priority areas for on-going surveillance are provided for Australia, which are used to guide current wild bird surveillance programs in Australia. Several practical considerations were also identified to improve future surveillance.

Long-term surveillance studies in wild birds in priority areas are recommended, which incorporate information on bird abundance, age, behaviour, breeding and movements, particularly for dabbling ducks. This is important to validate trends of LPAI prevalence, in understanding the main determinants for virus spread and persistence, and in predicting and managing future epizootics of HPAI in Australasia.

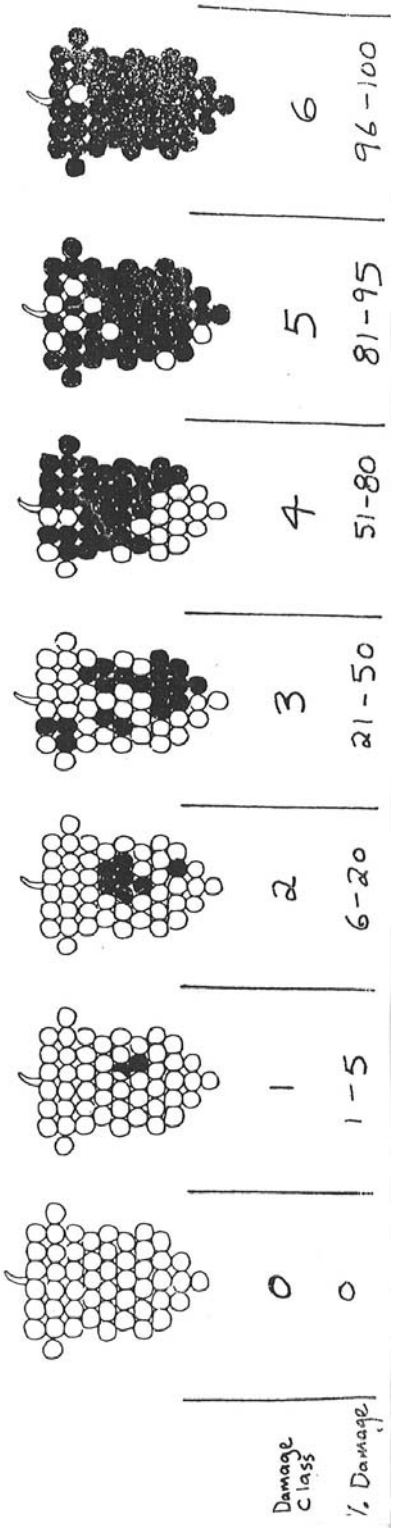
The case studies presented demonstrate that an understanding of a pest's ecology, efficient measures of impacts, and thorough evaluations of surveillance and management strategies are essential for effectively managing their economic, environmental and health-related impacts.

Future studies should include: (1) evaluations of the long-term benefits of management on pest bird populations, including optimal timing for control, and improved information on costs and benefits of controls over time; (2) assessing the environmental impacts of pests, including the use of genetic markers to quantify the extent of hybridisation between native and introduced species; and (3) long-term studies of diseases in wild birds in priority areas, which include the collection of ecological information. This is important in understanding and managing the impacts of avian diseases to human, animal and wildlife health.

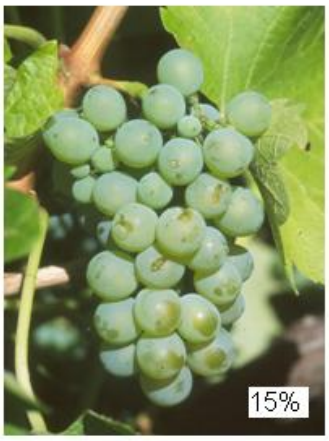
Appendix I: Examples of charts used to visually estimate bird damage

Bird Damage Assessment

Grape Variety:	*Red*	
	Cabernet Sauvignon	CB
	Shiraz	Sh
	Merlot	Mer
	Pinot Noir	Pin
	Cabernet Franc	CF
	Malbec	Mal
	White	
	Chardonnay	Char
	Sauvignon Blanc	SB
	Rhine Riesling	Reis
	Semmillon	Sem
	Verdelho	Ver
	Marsanne	Mar
	G Traminer	GT
Maturity:	Ripe	R
	Green	G
	Partially Ripe	PR
Exposed?:	Exposed	Y
	Not exposed	N
	Partially exposed	PE
Diseases:	Botryis bunch rot	Bot
	Black spot	BSp
	Powdery Mildew	PM
	Downy Mildew	DM
	Other Rot	Rot
	Insects	Insect
	Other/ Unknown	Other
Damage Type:	Pecked	P
	Missing	M



Bird Damage Assessment



Appendix II: Movements, abundance and distribution of birds with Australian and Asian distributions

Information in this table is derived from Blakers et al., 1984; Marchant and Higgins 1990, 1993; Higgins and Davies 1996; Higgins 1999; Kingsford and Norman 2002 and Higgins and Peter 2003. For consistency categories of Kingsford and Norman 2002 were used where possible. Movements are categorised as Sedentary (S); Nomadic (N); Partial Migrant (PM); Annual Migrant (AM). Abundance: Abundant (A); Locally Abundant (LA); Common (C); Locally Common (LC); UnCommon (UC); Vagrant (V); Rare (R). Ecological Information: Good; Moderate; Poor. Distribution: Continental, Northern (N), Eastern (E), Southern (S), Western (W), Pelagic and Coastal. Nomenclature follows Christidis and Boles 1994. Asterisks indicate avian influenza virus isolations.

Common name *Scientific name*	Aust-Asia movements	Movements	Timing of movement to Australia	Australian Abundance	Ecological information	Australian distribution
ANSERIFORMES						
Anatidae						
Plumed Whistling-Duck* *Dendrocygna eytoni*	Rare	N	Variable	LA	Moderate	N/SE
Wandering Whistling-Duck *Dendrocygna arcuata*	Regular	N	Variable	LA	Poor	N/NE
Radjah Shelduck *Tadorna radjah*	Rare	N	Variable	MC	Poor	N
Cotton Pygmy-goose *Nettapus coromandelianus*	Occasional	S	Variable	UC	Poor	NE
Green Pygmy-goose *Nettapus pulchellus*	Occasional	S	Variable	C	Poor	N
Pacific Black Duck* *Anas superciliosa*	Rare	S - N	Variable	A	Good	Continental
Northern Shoveler* *Anas clypeata*	Rare	AM	Aug, Sept	V	Poor	SW/SE
Grey Teal* *Anas gracilis*	Occasional	N	Variable	A	Good	Continental
Northern Pintail* *Anas acuta*	Rare	AM	Aug, Sept	V	Poor	SW
Garganey* *Anas querquedula*	Rare	AM - N	Aug, Sept	R - UC	Poor	N
Hardhead *Aythya australis*	Rare	N	Variable	LA - C	Moderate	Continental
Anseranatidae						
Magpie Goose *Anseranas semipalmata*	Regular	N	Variable	LA - C	Good	N
PROCELLARIIFORMES						
Procellariidae						
Streaked Shearwater *Calonectris leucomelas*	Regular	AM	May, Jun	UC	Poor	Pelagic
Wedge-tailed Shearwater* *Puffinus pacificus*	Occasional	AM	Aug, Sept	A	Poor	Pelagic
Flesh-footed Shearwater *Puffinus carneipes*	Occasional	AM	Sept, Oct	A	Poor	Pelagic

Appendix II. Continued

Common name *Scientific name*	Aust-Asia movements	Movements	Timing of movement to Australia	Australian Abundance	Ecological information	Australian distribution
Sooty Shearwater *Puffinus griseus*	Occasional	AM	Sept, Oct	MC	Poor	Pelagic
Short-tailed Shearwater *Puffinus tenuirostris*	Regular	AM	Aug, Sept	A	Poor	Pelagic
Hydrobatidae						
Matsudaira's Storm-Petrel *Oceanodroma matsudairae*	Regular	AM	Jul, Aug	UC	Poor	Pelagic
PELECANIFORMES						
Phaethontidae						
Red-tailed Tropicbird *Phaethon rubricauda*	Regular	N	Variable	LA	Poor	Pelagic
White-tailed Tropicbird *Phaethon lepturus*	Regular	N	Variable	UC	Poor	Pelagic
Sulidae						
Masked Booby *Sula dactylatra*	Regular	PM	Sep, Oct	LA	Poor	Pelagic
Red-footed Booby *Sula sula*	Regular	PM	Jun, July	LA	Poor	Pelagic
Brown Booby *Sula leucogaster*	Regular	PM	Variable	LA	Poor	Pelagic
Anhingidae						
Darter *Anhinga melanogaster*	Rare	N	Variable	MC	Moderate	Continental
Phalacrocoracidae						
Little Pied Cormorant *Phalacrocorax melanoleucos*	Occasional	N	Variable	A	Good	Continental
Little Black Cormorant *Phalacrocorax sulcirostris*	Occasional	N	Variable	A	Good	Continental
Great Cormorant* *Phalacrocorax carbo*	Rare	N	Variable	A	Good	Continental
Pelecanidae						
Australian Pelican *Pelecanus conspicillatus*	Occasional	N	Variable	LA - C	Good	Continental
Fregatidae						
Great Frigatebird *Fregata minor*	Regular	N	Variable	LA	Poor	Pelagic
Lesser Frigatebird *Fregata ariel*	Regular	N	Variable	LA	Poor	Pelagic
CICONIIFORMES						
Ardeidae						
White-faced Heron *Egretta novaehollandiae*	Occasional	S - N	Variable	C	Moderate	Continental
Little Egret *Egretta garzetta*	Occasional	S - N	Variable	C	Good	N/E/SE

Appendix II. Continued

Common name *Scientific name*	Aust-Asia movements	Movements	Timing of movement to Australia	Australian Abundance	Ecological information	Australian distribution
Eastern Reef Egret *Egretta sacra*	Occasional	S	Variable	C	Poor	Coastal
White-necked Heron *Ardea pacifica*	Rare	N	Variable	MC	Moderate	Continental
Pied Heron *Ardea picata*	Regular	S - N	Dec, Feb	LA-C	Moderate	N
Great Egret *Ardea alba*	Occasional	N	Variable	C	Good	Continental
Intermediate Egret *Ardea intermedia*	Rare	PM - N	Unknown	C	Moderate	N/E/SE
Cattle Egret *Ardea ibis*	Rare	PM	Unknown	LA -C	Moderate	SW/N/E/SE
Striated Heron *Butorides striatus*	Rare	S	Variable	LC	Poor	Coastal
Nankeen Night Heron *Nycticorax caledonicus*	Occasional	N - S	Variable	C	Poor	N/E/SE/W
Little Bittern *Ixobrychus minutus*	Occasional	AM	Aug	UC	Poor	SW/SE/E
Black Bittern *Ixobrychus flavicollis*	Unknown	S - N	Variable	LC - UC	Poor	N/E
Threskiornithidae						
Glossy Ibis* *Plegadis falcinellus*	Occasional	N	Variable	LA	Moderate	Continental
Australian White Ibis *Threskiornis molucca*	Occasional	S - N	Variable	LA - C	Good	Continental
Straw-necked Ibis *Threskiornis spinicollis*	Occasional	S - N	Variable	C	Good	Continental
Royal Spoonbill *Platalea regia*	Occasional	S - N	Variable	C	Moderate	N/E/SE/W
GRUIFORMES						
Rallidae						
Spotless Crake *Porzana tabuensis*	Unknown	S - N	Unknown	C - UC	Poor	SW/SE
White-browed Crake *Porzana cinerea*	Unknown	S	Unknown	LC	Poor	N
Purple Swamphen *Porphyrio porphyrio*	Unknown	N	Unknown	C	Moderate	SW/E
CHARADRIIFORMES						
Scolopacidae						
Latham's Snipe *Gallinago hardwickii*	Regular	AM	Aug	LC	Poor	SE/E
Swinhoe's Snipe *Gallinago megala*	Regular	AM	Aug	MC	Poor	SE/E
Black-tailed Godwit *Limosa limosa*	Regular	AM	Sept, Oct	MC	Moderate	Continental

Appendix II. Continued

Common name *Scientific name*	Aust-Asia movements	Movements	Timing of movement to Australia	Australian Abundance	Ecological information	Australian distribution
Bar-tailed Godwit *Limosa lapponica*	Regular	AM	Sept, Oct	C	Moderate	Continental
Little Curlew *Numenius minutus*	Regular	AM	Oct	LA - MC	Moderate	N
Whimbrel *Numenius phaeopus*	Regular	AM	Sept, Oct	MC	Poor	Coastal
Eastern Curlew *Numenius madagascariensis*	Regular	AM	Sept, Oct	MC	Moderate	Coastal
Marsh Sandpiper *Tringa stagnatilis*	Regular	AM	Oct, Nov	MC - UC	Moderate	Continental
Common Greenshank *Tringa nebularia*	Regular	AM	Oct, Nov	C	Moderate	Continental
Wood Sandpiper *Tringa glareola*	Regular	AM	Jul, Oct	MC - UC	Moderate	Continental
Terek Sandpiper *Xenus cinereus*	Regular	AM	Sept, Oct	MC - UC	Moderate	Coastal
Common Sandpiper *Actitis hypoleucos*	Regular	AM	Sept, Oct	MC - UC	Moderate	Continental
Grey-tailed Tattler *Heteroscelus brevipes*	Regular	AM	Sept, Oct	C	Poor	Coastal
Wandering Tattler *Heteroscelus incanus*	Regular	AM	Nov	UC	Poor	Coastal
Ruddy Turnstone* *Arenaria interpres*	Regular	AM	Sept, Oct	MC	Moderate	Continental
Asian Dowitcher *Limnodromus semipalmatus*	Regular	AM	Sept, Oct	R	Poor	Coastal
Great Knot *Calidris tenuirostris*	Regular	AM	Sept, Oct	LA - UC	Moderate	Coastal
Red Knot* *Calidris canutus*	Regular	AM	Sept, Oct	C	Poor	Coastal
Sanderling *Calidris alba*	Regular	AM	Sept, Oct	MC	Poor	Coastal
Red-necked Stint* *Calidris ruficollis*	Regular	AM	Sept, Oct	A - C	Moderate	Continental
Long-toed Stint *Calidris subminuta*	Regular	AM	Aug	UC	Moderate	Continental
Pectoral Sandpiper *Calidris melanotos*	Occasional	AM	Oct, Nov	UC	Poor	Continental
Sharp-tailed Sandpiper *Calidris acuminata*	Regular	AM	Sept, Oct	C	Moderate	Continental
Curlew Sandpiper *Calidris ferruginea*	Regular	AM	Sept, Oct	A - C	Moderate	Continental
Broad-billed Sandpiper *Limicola falcinellus*	Regular	AM	Sept, Oct	MC - UC	Poor	Coastal

Appendix II. Continued

Common name *Scientific name*	Aust-Asia movements	Movements	Timing of movement to Australia	Australian Abundance	Ecological information	Australian distribution
Jacanidae						
Comb-crested Jacana *Irediparra gallinacea*	Rare	S	Variable	C	Poor	N/NE
Charadriidae						
Grey Plover *Pluvialis squatarola*	Regular	AM	Sept, Oct	MC	Moderate	Coastal
Lesser Sand Plover *Charadrius mongolus*	Regular	AM	Sept, Oct	MC	Poor	Coastal
Greater Sand Plover *Charadrius leschenaultii*	Regular	AM	Sept, Oct	UC	Moderate	Coastal
Oriental Plover *Charadrius veredus*	Regular	AM	Oct, Nov	LA - UC	Poor	N/SE
Masked Lapwing *Vanellus miles*	Unknown	S – N	Unknown	C	Good	N/E/SE
Glareolidae						
Oriental Pratincole *Glareola maldivarum*	Regular	AM	Nov	LA - UC	Moderate	N/W
Australian Pratincole *Stiltia isabella*	Regular	AM	Aug, Sep	C - UC	Moderate	Continental
Laridae						
Lesser Crested Tern *Sterna bengalensis*	Unknown	S	Unknown	C	Poor	Coastal
Crested Tern *Sterna bergii*	Unknown	S	Unknown	C	Poor	Coastal
Roseate Tern *Sterna dougallii*	Regular	S – N	Variable	C - UC	Poor	Coastal
Black-naped Tern *Sterna sumatrana*	Unknown	S – N	Unknown	LC	Poor	Coastal
Common Tern* *Sterna hirundo*	Regular	AM	Sept, Oct	MC	Good	Coastal
Little Tern *Sterna albifrons*	Regular	AM	Variable	MC - UC	Moderate	Coastal
Bridled Tern *Sterna anaethetus*	Regular	AM	Jan, Feb	C	Poor	Coastal
Sooty Tern* *Sterna fuscata*	Regular	AM	Variable	A	Moderate	Coastal
Whiskered Tern *Chlidonias hybridus*	Regular	AM – N	Sept, Oct	A - C	Moderate	Continental
White-winged Black Tern *Chlidonias leucopterus*	Regular	AM – N	Sept, Oct	MC - UC	Poor	Continental
COLUMBIFORMES						
Columbidae						
Superb Fruit-Dove *Ptilinopus superbus*	Regular	N	Sept, Oct	UC	Moderate	E
Pied Imperial-Pigeon *Ducula bicolor*	Regular	AM	Aug, Sept	C - MC	Moderate	N

Appendix II. Continued

Common name *Scientific name*	Aust-Asia movements	Movements	Timing of movement to Australia	Australian Abundance	Ecological information	Australian distribution
CUCULIFORMES						
Cuculidae						
Brush Cuckoo *Cacomantis variolosus*	Regular	AM	Sept, Oct	MC	Moderate	N/E
Common Koel *Eudynamys scolopacea*	Regular	PM	Sept, Oct	MC	Moderate	N/E
Channel-billed Cuckoo *Scythrops novaehollandiae*	Regular	PM	Sept, Oct	MC	Moderate	N/S
CORACIIFORMES						
Halcyonidae						
Buff-breasted Paradise-Kingfisher *Tanysiptera sylvia*	Regular	AM	Sept, Oct	UC	Moderate	NE
Forest Kingfisher *Todiramphus macleayii*	Regular	AM	Sept, Oct	C	Good	NE/E
Sacred Kingfisher *Todiramphus sanctus*	Regular	AM	Sept, Oct	C	Good	Continental
Meropidae						
Rainbow Bee-eater *Merops ornatus*	Regular	AM	Sept, Oct	C	Good	Continental
Coraciidae						
Dollarbird *Eurystomus orientalis*	Regular	AM	Sept, Oct	MC	Good	N/E
PASSERIFORMES						
Meliphagidae						
Brown-backed Honeyeater *Ramsayornis modestus*	Occasional	S	Sept, Oct	LC	Poor	NE
Dicruridae						
Spangled Drongo *Dicrurus bracteatus*	Regular	S – AM	Sept, Oct	C	Moderate	NE
Campephagidae						
Cicadabird *Coracina tenuirostris*	Regular	AM	Sept, Oct	UC - MC	Poor	NE
Oriolidae						
Olive-backed Oriole *Oriolus sagittatus*	Occasional	PM - AM	Sept, Oct	C - MC	Moderate	N/E
Sturnidae						
Metallic Starling *Aplonis metallica*	Regular	AM	Sept, Oct	C - LC	Moderate	NE

Appendix III: Analyses* of Anseriformes movements and abundance and avian influenza surveillance data.

Response variable →	**Ln (Anseriformes distance moved[1])**		**Ln (Anseriformes abundance[2])**		**LPAI prevalence[3] Global**		**LPAI prevalence[3] Australia**	
Dependent variable ↓	Wald statistic	*P*	Wald statistic	*P*	Wald statistic	*P*	Wald statistic	*P*
Continent (Australia, overseas)	NA	NA	NA	NA	F(1, 174) = 8.630	0.003	NA	NA
Functional group[4]	NS	NS	NA	NA	F(14, 208) = 3.236	0.0001	F(14, 550) = 2.079	0.009
Ln (Anseriformes abundance)	NA	NA	NA	NA	NA	NA	F(1, 536) = 15.640	<0.0001
Species	NS	NS	NA	NA	NA	NA	NA	NA
Sex	NS	NS	NA	NA	NA	NA	NA	NA
Age	NS	NS	NA	NA	NA	NA	NA	NA
Season	NS	NS	NA	NA	NS	NS	NS	NS
Seasonal rainfall classification[5]	F(6, 348) = 6.129	<0.0001	F(5, 13 424) = 55.8	<0.0001	NA	NA	F(5, 533) = 3.377	0.003
Availability of permanent water[6] (km^2 per 1/4° grid)	NS	NS	F(1, 13 424) = 265.8	<0.0001	NA	NA	F(1, 579) = 30.220	<0.0001
Percentage of permanent water[6] (% of 1/4° grid)	NS	NS	NS	NS	NA	NA	NA	NA
Annual rainfall[7]	NS	NS	NA	NA	NA	NA	NA	NA
Mean annual rainfall[7]	F(1, 443) = 9.976	0.0017	NA	NA	NA	NA	NA	NA
Seasonal rainfall[7]	NS	NS	NA	NA	NA	NA	NA	NA

Response variable →	**Ln (Anseriformes distance moved[1])**		**Ln (Anseriformes abundance[2])**		**LPAI prevalence[3] Global**		**LPAI prevalence[3] Australia**	
Dependent variable ↓	Wald statistic	*P*	Wald statistic	*P*	Wald statistic	*P*	Wald statistic	*P*
Rainfall in the previous two months[7]	NS	NS	NA	NA	NA	NA	NA	NA
Rainfall in the previous three months[7]	NS	NS	NA	NA	NA	NA	NA	NA
Functional Group:Continent	NA	NA	NA	NA	F(13, 628) = 5.521	<0.0001	NA	NA
Ln (Anseriformes abundance): Availability of permanent water	NA	NA	NA	NA	NA	NA	F(1, 579) = 15.720	<0.0001
Other interactions	NS	NS	NS	NS	NS	NS	NS	NS
Random variables	Site, sampling event		NA		Site, sampling event, species		Site, sampling event, species	

* Generalised linear mixed models and analysis of variance were conducted using the *asreml* package (Butler et al., 2009) under R (R Core Team, 2013). NA indicates the variables that were not included in the analyses, where data was not available and NS indicates variables that were not significant at the $P = 0.05$ level that were dropped sequentially.

[1]Distances moved for Anseriformes were estimated as kilometres travelled from the capture location, using banding data from the Australian Bird and Bat Banding Scheme (www.environment.gov.au/biodiversity/science/abbbs/, accessed 1 January 2009)

[2]The abundance of Anseriformes was estimated from the relationship between the number of birds and number of species per observation (see Results 7.3) using Birds Australia's 20-min, 2-ha search for species reporting rate and bird-count data (Barrett et al., 2003).

[3]LPAI prevalence was estimated as the proportion of samples (cloacal, oropharyngeal or faecal swabs) from wild birds that were positive to Influenza A (via Polymerase chain reaction and / or virus isolations)

[4]Functional group were groupings of species based on behaviour and feeding habits (after Kingsford, 1991 and Roshier et al., 2002)

[5]Seasonal rainfall classification: a Bureau of Meteorology classification, identifying summer-dominant, summer, uniform, winter, winter-dominant, or arid rainfall

[6]Permanent water Geoscience Australia and National Water Commission data on rivers, dams and inland water

[7]Rainfall data from the Bureau of Meteorology

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